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Article

Application of an Expanded Sequestration Estimate to the Domestic Energy Footprint of the Republic of Ireland

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Abstract: The need for global comparability has led to the recent standardization of ecological footprint methods. The use of global averages and necessary methodological assumptions has questioned the ability of the ecological footprint to represent local or national specific concerns. This paper attempts to incorporate greater national relevancy by expanding the sequestration estimate used to calculate the annual carbon footprint of domestic Irish energy use. This includes expanding existing study boundaries to include additional carbon pools such as the litter, dead and soil pools. This generated an overall estimate of 4.38 tonnes of carbon per hectare per year (t C/ha/yr), resulting in an ecological footprint estimate of 0.49 hectares per capita (ha/cap) The method employed in this paper also incorporated the potential role of grassland as a carbon sink. The caveat that the resultant value is dependent on the choice of study boundary is discussed. Including the lateral movement of carbon embodied in farm products (effectively placing the boundary around the farm gate) reduces the estimate of grassland carbon sequestration by approximately 44% to 1.82 t C/ha/yr. When a footprint calculated using an overall sequestration estimate (based on the distribution of Irish grassland and forestry) is translated into global hectares (gha), the standardized value is reduced by 35%.

Keywords: sequestration; carbon footprint; forest; grassland

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1. Introduction

The recent Footprint of Nations World Wildlife Fund [1] shows that Ireland's energy footprint is the single largest contributor to the overall footprint. Current ecological footprint methods account for impacts of fossil energy use through estimating the area of forestry required to sequester the associated emissions of carbon dioxide. Forest areas of global average productivity are used for this purpose, and are translated into as area of overall global average productivity, using units termed global hectares. As the forest area allocated globally for carbon sequestration is negligible, there is little capacity for sufficient land area to be made available. This highlights the finite nature of natural capital and reinforces that only a reduction in energy demand can reduce the ecological deficit. Translating diverse categories of resource use into a single metric allows for ready comparison between the overall supply and demand of bio-capacity, facilitating effective communication through a single unit. However, any method which focuses on global averages may fail to convey the complexities inherent in resource provision. Kitzes *et al.* [2] suggest a number of means of improving the ecological footprint. These include incorporating other land types into the energy sequestration estimate, as well as calculating footprints using local hectares as opposed to global hectares (often termed 'actual' footprinting). This study presents a number of alternatives for the inclusion of specific Irish conditions into the energy footprint method. In essence, these options involve expanding the current system boundaries of the carbon footprint method to include additional carbon sinks such as the forest litter and deal wood pool as well as forest and grassland soil. The main objective of this study is add to the debate began in [2], specifically the extent to which footprint results are dependent on choices of land type identified as a carbon sink.

2. Methodological Option for Including Additional Carbon Sinks into Ecological Footprint Methodology

2.1. Tree Increment Pool

The process of tree growth results in a net carbon sink as the carbon dioxide (CO_2) fixed through photosynthesis exceeds the CO_2 emitted through respiration. Forest growth takes place over three stages, representative of growth throughout the life of a tree as well as reflecting the growth within a single season [3]. The variability in carbon sequestration—both annually and seasonally—will depend on a variety of environmental factors such as the availability of light, temperature, drought or soil fertility. The juvenile growth phase represents the period of accelerating growth and follows an exponential curve as described by Long and Smith [4]. Once growth reaches maximum, increments follow a general logistic curve [5]. This curve includes a period of mature growth where the growth rate is constant followed by the senescent phase which demonstrates a period of declining growth. Although this relationship between tree volume and age is basically sigmoidal, the cumulative growth curve (CGC) may be affected by climatic factors and changes in silviculture management. Brack and Wood [3] list some of the main determinants of forest growth such as initial spacing and treatment. However, it is important not to confuse the growth of a single tree with the growth of a forest in general, which is dependent on the interaction of a wide variety of environmental determinants. Current annual increment (CAI) and mean annual increment (MAI) are the conventional expressions of an increase in tree matter. CAI measures the present increase in biomass while periodic current annual increment (PCAI) measures the average CAI over a given timescale. MAI is calculated as the total cumulated production divided by stand age, reflecting the average increment over the entire specified age. Provided CAI exceeds MAI, an increase in increment is expected, as the average should rise in the future. After the CAI reaches its peak, the MAI curve will appear to plateau, reaching a maximum point where both curves intersect. Beyond this, the MAI will decrease but at a slower rate than the CAI. The age of intersection is important for commercial forest planning as it represents the optimal point for felling, as thereafter continued growth will provide a lower than average yield [3]. In estimating an 'actual footprint' the specific distribution of tree species and ages is necessary to generate an Irish specific sequestration estimate and would not be meaningful otherwise. It could be argued that this method does not represent a strict methodological difference but rather represents a utilization of different data. As well as greater specificity, this method will reflect recent forestry management, such as reliance on conifers. However, it should be mentioned that since an ecological footprint represents the optimal area preserved for a single purpose (in this case the sequestration of carbon), it is assumed that no thinning of forest stocks take place.

2.2. The Litter, Dead and Soil Pools

The litter pool is currently disregarded within the carbon sequestration estimate used within ecological footprinting [1]. However, the accumulation of litter on the forest floor (assuming the area is not significantly disturbed) can increase the overall forest carbon pool. Litter accumulation and subsequent decomposition is an important mechanism for the return of nutrients such as P and N to the forest nutrient cycles. The rate and extent of litter decomposition is determined by a number of factors. However, litter decomposition is most closely determined by the activity of microbial communities and soil fauna. Microbial activity is influenced by prevailing climatic conditions and the bio-physical nature of the litter [6]. Other factors, such as clear cutting, have been suggested as increasing litter decomposition, as the forest floor is generally warmer and wetter following clear felling. However, other authors suggest that the effect of clear felling in practice depends on climatic factors, as wind exposure may dry litter and counteract the effect of increased sunlight and moisture [7]. Also overlooked is the function of the dead wood pool in relation to carbon sequestration. As with litter, the sequestration rate of the (unused) dead pool is determined by the net gain (accumulation minus decomposition) of carbon. The capacity of forest soils to sequester carbon is governed by the residency and accumulation of a resistant slowly decomposable C pool. Climatic factors such as soil temperature and rainfall can determine microbial activity, which will ultimately determine loss of soil organic carbon (SOC) through respiration [8]. Because of disturbance, new forestry plantations will likely be net sources of carbon emissions for a number of years until in-growth and detritus begin to return carbon to the soil [9]. Once a forest area has been stabilized, practices such as ensuring a continuous canopy cover and attempting to replicate the natural patterns of disturbance will assist in maintaining both a high wood yield and an increased SOC pool [10]. Although being less than the carbon stored in trees, the additional forest sinks are significant when viewed collectively. Hence, this method may have significant implications for footprinting in general.

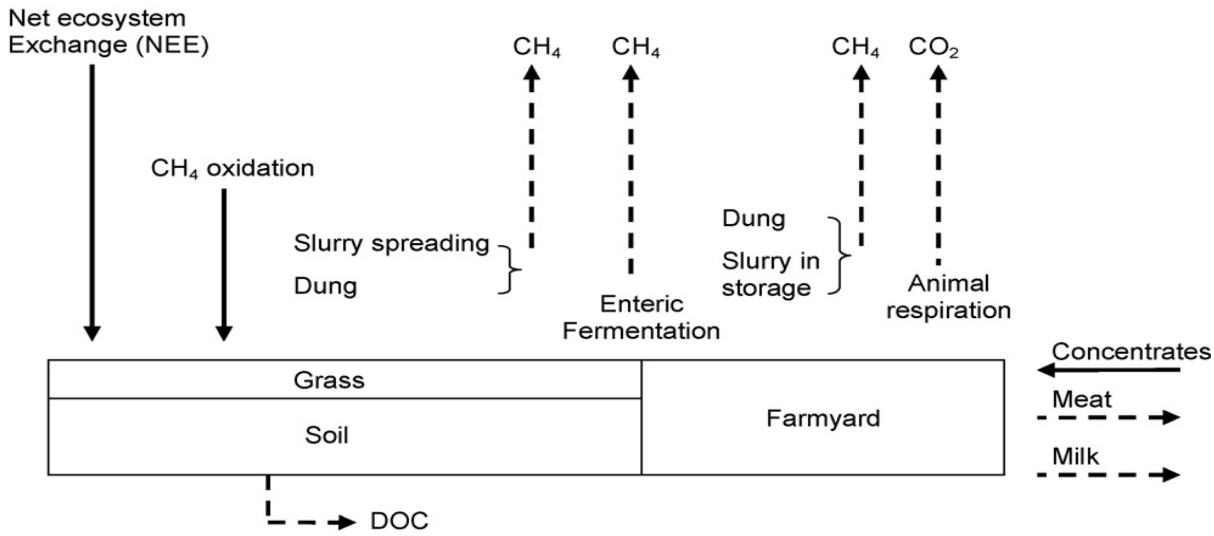
2.3. Grassland Sequestration

In contrast to forestry, soil comprises the vast majority of 'labile' carbon in a grassland ecosystem, and therefore, the sequestration service of grassland is a service provided by the soil. Jones and Donnelly [11] provide an account of sequestration within temperate grassland areas. Soil organic matter (SOM) accrues within grassland soil through the decomposition of roots, senescent leaves and stems within the soil. Within most temperate grasslands, the majority of root biomass occurs within the top 30 cm of the soil. However, the process of root growth, death and decomposition occur simultaneously and at different rates, and are determined by species composition and climatic conditions. The death and decomposition of root material is not the only means by which carbon is added to soils: rhizodeposition is also important, resulting from mucilage production, root exudation, and fragmentation from living roots. Because of the relative quantity of root matter within the surface soil layers, it is difficult to differentiate between the processes of CO_2 production and carbon cycling that originates from plant matter, and those which are derived from microbial activity [12]. The SOC content is considered to reside within discrete pools. A considerable fraction of SOM is enclosed by clay minerals within soil aggregates; therefore, it remains physically protected through processes such as chemical binding with soil minerals. Structural protection is also provided by soil aggregates, which provide a physical impediment for microbes, enzymes and organic substrates [13]. Soil structure also plays an important role as organic matter may be located within pores that are too small to allow access for fungi and bacteria. Soil aggregates themselves are held together by fungal hyphae, roots, microbial debris and polysaccharides, such that an increase in any of these fractions may promote intermediate SOM stability. Schulze *et al.* [14] summarize the processes that are relevant to carbon sequestration within a temperate European (intensively managed) grassland, and illustrate that the actual changes in SOC may be dwarfed by the size of the carbon fluxes entering and leaving the soil boundary.

Expanding the carbon sequestration estimate away from using forestry as the reference land type has fundamental ramifications for footprint results and methodology. However, in some regards this alternative may be potentially more arbitrary as it poses a crucial methodological question. This is related to the discussion of single land use and how the energy footprint functions as a theoretical or notional measurement. The conventional energy footprint is treated as an additional area that must be maintained to offset current CO_2 emissions, commercial forest areas are not suitable as they are at best carbon neutral. Treating grassland in the same manner would mean incorporating net ecosystem exchange (NEE), which equates to net primary production minus heterotrophic respiration and represents the fluxes of carbon experienced by an area of grassland. By contrast, Byrne *et al.* [15] calculates a farm-based sequestration estimate, which treats grassland as operating within a system and incorporates the lateral movement of carbon embodied in farm commodities across the farm boundary. However the estimate of NEE calculated in [15] will include the emissions due to cattle respiring while grazing (Figure 1).

The fact that commercial and productive grassland can function as a carbon sink may seem to negate the presumption of a single land-use, however that is by no means certain. For that reason, two estimates of grassland sequestration were adopted in this study: one based on NEE and one where the movement of carbon is included. The integration of both grassland- and forest-based estimates represents a further methodological decision. In this study, overall estimates are generated based on the relative area of grassland and forestry. However, as Ireland possesses a large area of very productive grassland, this may not be suitable for application in other countries and a simple average may be preferable.

Figure 1. Schematic of the farm-based carbon balance, taken from [15]. Note: slurry refers to manure stored specifically for use as a fertilizer.



3. Calculation Methods

3.1. Calculating a Carbon Sequestration Estimate Based on Forest Increment

The national inventory report [16] includes a detailed estimate of the carbon sequestered through Irish forests, and assumes the land will be retained for the specific purpose of carbon capture. The national inventory report uses data on both the species and age (in intervals of 10 years), and distribution of the national forest estate between 2004 and 2006 [16]. However, in order to estimate forest growth increment, the average yield of each species at a given age must be estimated. Staff at Coillte, the Irish sponsored forestry company Lynch [17], provide yield class estimates for a number of conifer species, while Magner [18] provide estimates on the likely yield classes of Irish broadleaves, with additional yield class estimates taken from Horgan *et al.* [19] and Joyce *et al.* [20]. Staff at Coillte also provided a copy of the British Forestry Commission (BFC) yield program [21], which is a software framework for modeling forest growth and obtaining yield data.

$$CS = V \times D \times ABEF \times (1 + R:S) \times CF \qquad (1)$$

In Equation 1, the annual carbon sequestration due to tree increment (CS) measured in tonnes of carbon per hectare per year is estimated. V refers to stem wood volume (PCAI estimates were used); D refers to basic density (dry kg/m^3); ABEF is the ratio between total above ground biomass and stem wood: R:S refers to the root to shoot ratio and effectively measures the ratio between above-ground and below-ground biomass; and CF refers to the carbon content of tree biomass, which is approximately 50% for most species. Tobin *et al.* [22] provide data for use on Sitka spruce, while Lehtonen *et al.* [23] and Levy *et al.* [24] provide data for other species.

3.2. Carbon Accumulation in the Litter, Dead and Soil Pools

While tree biomass represents the main carbon sink within forestry it does not represent the total carbon sink. Net ecosystem productivity (NEP) for a forest area can be crudely simplified as the change in carbon stocks in biomass, above ground litter, deadwood and soil. The annual carbon gains through litter fall were calculated for each tree species at each individual year. Equation 2 forms the basis for this calculation.

$$G_n = Vn \times D \times ABEF \times CF \times 0.2 \times (9.6/100) \times 1.67 \tag{2}$$

In Equation 2, G_n refers to the gross carbon gained through litterfall accumulation and is measured in tonnes of carbon per hectare per year. The value ‘0.2’ refers to the annual conifer leave turnover (1 for broadleaves). The value ‘9.6’ refers to the percentage of above ground biomass allocated to leaves. As age increases, the leaf biomass remains relatively constant while stem wood increases; therefore, this value is reduced to 4% beyond 17 years. The value ‘1.67’ represents an attempt to account for the wood fraction of litter fall, which was found to comprise up to 40% of coniferous forest floors: this value was changed to 21% for broadleaf floors as the wood fraction was considerably less [25]. The loss of carbon due to decomposition is more complicated due to the fact that while litter accumulation happens on an annual basis (and is therefore easily identifiable), decomposition of the litter fall from previous years occurs continually. Carbon losses due to decomposition (L_n) were estimated as a function of leaf carbon litter gains from previous years. As with Equations 1 and 2 above, it is measured in tonnes of carbon per hectare per year.

$$L_n = \sum_{1}^{15} G_{n-(n-1)} \times (1-k)^{(n-1)} \times k \tag{3}$$

In Equation 3, an attempt was made to account for the compound nature of litter decomposition within a given year. G_n refers to carbon gains within a given year, estimated using Equation 2 and ‘k’ refers to litter turnover or decay rate, assumed to be 0.14 yr^{-1} [26]. Effectively, the carbon loss for any given year, n, is estimated as the gain of the current year multiplied by the turnover rate plus the net carbon gain from the fall of previous years multiplied by litter turnover. Because this is a compound method, the actual loss from a given year’s litterfall will decrease with each subsequent year. If the amount available for decomposition decreases by 14% each year, then 90% of a given aboveground leaf litter fall is decomposed within 15 years. A similar method was adopted for deadwood, using whole tree biomass as opposed to above ground biomass, whereby an average annual tree mortality rate was estimated by dividing the volume of recently dead trees by the total growing stock volume; both estimates taken from [16]. This resulted in an annual mortality estimate of 0.35%. Losses due to the decomposition of deadwood were calculated in the same manner as in Equation 3. A decay rate of 0.1 yr^{-1}, as used within the Australian greenhouse gas (GHG) inventory report, taken from Mackensen and Bauhus [27], was applied. The compound nature of turnover means that 23 years are required before 90% of dead tree matter is decomposed.

Estimates of soil stock change due to afforestation were taken directly from relevant literature. The 2008 National GHG inventory report [28] suggests that newly afforested organic soils emit 16 tonnes of carbon per hectare per year (t C/ha/yr) over a transition period of four years [28].

Because newly afforested areas are represented within the national forest inventory as the area planted less than 10 year ago, this emission factor was assumed to apply to 40% of the planted area and therefore was reduced to 1.6 t C/ha/yr. This new emission factor was applied to only 30% of forested area under 10 years of age, as organic soils are though to represent 30% of afforested areas planted from 2003 onwards. For older forest stands, a uniform sequestration rate was applied. This was taken from Reidy *et al.* [29], who estimate that soil within selected Irish stands may sequester between 0.2 and 2.3 t C/ha/yr. Because of the variable nature of respiration rates (and subsequently, soil sequestration rates), a conservative estimate of 0.2 t C/ha/yr was applied to the soil of all forest plantations older than 10 years. This is due to the possibility that older forest soils may be approaching an equilibrium state.

3.3. Carbon Sequestration in Irish Grasslands

The carbon sequestration rates of Irish grassland areas were taken from Byrne *et al.* [15]. It is assumed that the area under examination (a dairy and beef farm in County Cork) is typical to other areas of pastureland in Ireland. The carbon sequestration estimate is calculated as part of a farm carbon budget including the lateral movement of carbon (the farm gate is viewed as the system boundary) embodied in farm products such as milk. It is assumed that carbon that leaves the pasture area (and subsequently the farm gate) should be excluded from the carbon pool. By contrast, NEE represents gross primary productivity (GPP) minus ecosystem respiration, which excludes the lateral movement of carbon. However, the value of NEE estimated by the eddy covariance tower also includes the CO_2 emissions released due to respiration and decomposition of field manure. This means that the value of NEE shown in [15] is an underestimate of actual carbon sequestration. While data on emissions due to field manure were unavailable, it is likely to be negligible in comparison to respiration estimates. In order to estimate emissions due to metabolic respiration, the emission rates applied to animals while indoors (6.75, 5.2 and 3.5 kg CO_2 per day for dairy animals, bullocks and calves, respectively) were applied to the herd. The value of NEE was corrected for by using herd distribution and stocking rates during the grazing period (which lasts from the end of March to Mid-October). As the farm in [15] is situated on mineral soil, it is not representative of area on organic soil. The 2008 National Inventory Report calculation spreadsheet [28] estimates the area of drained organic soils and applies an average emission factor of 0.25 t C/ha/yr. As this fails to include carbon movement, the above value was substituted for the NEE estimate within the carbon budget calculated by [15], and was used unaltered within the NEE based calculation.

4. Results

4.1. Forest Sequestration

Table 1 provides an example of the annual sequestration estimate calculated here for specific tree species. A more complete list is available as supplementary information. Based on the distribution of species and age groupings, an average hectare of Irish forestry was estimated to sequester 4.38 t C/ha/yr. Sitka spruce represented the largest contributor to the overall estimate. Black *et al.* [30] report that the average carbon sequestration rate through the lifetime of an examined Irish stand was between 4 and 8 t C/ha/yr. Please note that the values expressed here are measured in Irish hectares. Translating these estimates into global hectares is addressed in the discussion.

Table 1. Example of annual forest carbon sequestration estimates calculated for the indicated tree species.

Age (years)	**1,000 ha**	**Increment**	**Litter**	**Dead**	**Soil**
		t C/ha/yr			
Sitka spruce					
1–10	112.14	2.25	0.16	0.00	–0.59
11–20	120.90	7.13	0.26	0.18	0.20
21–30	38.68	6.14	0.25	0.43	0.20
31–40	40.93	6.09	0.44	0.53	0.20
41–50	14.15	4.40	0.40	0.53	0.20
51+	1.02	2.85	0.44	0.47	0.20
Norway spruce					
1–10	11.55	0.41	0.04	0.00	–0.59
11–20	6.19	3.02	0.12	0.04	0.20
21–30	1.24	8.60	0.30	0.15	0.20
31–40	2.95	9.25	0.52	0.23	0.20
41–50	3.25	7.79	0.58	0.25	0.20
51+	0.77	5.52	0.60	0.23	0.20
Scots pine					
1–10	2.22	0.41	0.04	0.00	–0.59
11–20	1.11	1.39	0.07	0.02	0.20
21–30	0.19	2.47	0.13	0.11	0.20
31–40	0.64	4.62	0.26	0.13	0.20
41–50	0.61	4.56	0.31	0.14	0.20
51+	2.57	2.82	0.33	0.13	0.20
Other Pine					
1–10	10.59	0.78	0.06	0.06	–0.59
11–20	15.25	3.12	0.15	0.15	0.20
21–30	23.81	5.40	0.23	0.23	0.20
31–40	9.78	4.68	0.35	0.35	0.20
41–50	3.11	3.96	0.35	0.35	0.20
51+	1.07	2.88	0.35	0.35	0.20

The carbon sequestered due to tree biomass is largely similar to those provided elsewhere [21]. However, the Intergovernmental Panel on Climate Change (IPCC) default sequestration values for leaf litter during the whole transition period of broad leaf and evergreen forests within cold temperate moist areas is estimated to be 0.3 and 0.5 t C/ha/yr, respectively, by Penman *et al.* [31], but were calculated within this study as 0.36 and 0.25 t C/ha/yr, respectively. As can be seen, the average carbon gained through coniferous litter accumulation is approximately half the default IPCC estimate. This may be explained by the age distribution of the national forest estate, as 70% of coniferous stands are below 20 years of age, as well as the specific decomposition rate chosen.

4.2. Estimating Grassland Sequestration

Byrne *et al.* [15] provide an estimate of carbon sequestered within an area of productive Irish grassland (Table 2). This estimate followed a farm survey (which relates to two farmed grassland areas in County Cork: farms A and B) and represents the amount of carbon retained by a pasture-based farm area when the movement of embodied carbon through the farm boundary is taken into account. However, the area examined in Byrne *et al.* [15] is not a drained organic/peaty soil. In order to compensate for the decomposition of organic matter, the estimate of NEE was replaced by the net emission from such soils [28]. The methane oxidation rate was replaced by a value which represents an average of grassland and bog areas, taken from Smyth *et al.* [32]. These estimates were applied to the pasture area provided by the Central Statistics Organization (CSO). The area of organic soils under pasture in 2007 was taken from [28]. This value was subtracted from the total pasture area and the net carbon balance of a peaty grassland area (−0.25 t C/ha/yr) was applied. Averages of the values for farm A and B were used to estimate the other categories within the organic farm. As this value does not include the effect of outdoor respiration, the value for animal respiration generated for farm B reflects the entire year as opposed to the period indoors.

Table 2. Farm-based carbon sequestration estimate relating to two farmed grassland areas in County Cork, Ireland: farms A and B, taken from [15] and augmented.

	Farm A	**Farm B**	**Organic (estimated)**
Carbon inputs	**t C/ha/yr**		
Net ecosystem (NEE)	2.9	2.9	−0.25
Concentrates	0.4	0.68	0.54
CH_4 oxidation	0.0015	0.0015	0.0028
Inputs sub Total	3.3015	3.5815	0.29282
Carbon outputs			
Milk	–0.21	–0.31	−0.26
Meat	–0.02	–0.02	–0.02
Enteric fermentation	–0.11	–0.12	–0.115
CH_4—Dung in farmyard	–0.001	–0.001	–0.001
CH_4—Dung in field	–0.0005	–0.0005	–0.0005
CH_4—Slurry spreading	–0.08	–0.06	–0.07
DOC (in stream)	–0.1	–0.1	–0.1
CH_4—Slurry in storage	–0.0001	–0.0001	–0.0001
Animal respiration	–0.73	–0.82	–1.3281
Outputs Sub Total	–1.2516	–1.4316	–1.3416
Net balance	**2.0499**	**2.1499**	**–1.6018**

Based on the herd distribution and respiration rates published in [15], it was suggested that cattle respire 0.53 and 0.58 t C/ha/yr during outdoor periods for farms A and B, respectively. This value was averaged and added from the existing NEE and it is assumed to cancel out the reduction of NEE due to animal respiration while grazing. The resultant values are hoped to approximate the sequestration capacity of natural grassland. A modified NEE value should be viewed with a caveat—as there is still possibility that extraneous carbon emissions (such as those relating to fossil fuel combustion) are detected by eddy covariance equipment, although the pasture site was a considerable distance from the main farm. The change in animal diet associated with indoor housing (such as the use of dried food) will, due to changes in digestibility, more readily alter the enteric fermentation estimate; however this has presumably been taken into account in [15]. Similarly, the distribution of different animal types, time outdoors, and herd size is explicitly detailed in [15] and corresponds with conventional practices of Irish pasture management. Nevertheless, the assumption that indoor respiration rates are suitable proxies for outdoor estimates may be questioned. Table 3 demonstrates the effect of lateral carbon movement as well as outdoor respiration on the sequestration estimate for both soil types.

Table 3. Net ecosystem (NEE) and farm based sequestration estimate for grassland.

	Area	**T C/ha/yr**	
	1,000 ha.	Farm	NEE
Mineral soil	3,553	2.1	3.45
Organic soil	289	–1.6	–0.25
Average		1.82	3.17

The inclusion of lateral carbon movement within a farm balance reduces the sequestration estimate (NEE) in Table 2 by approximately 44%. Animal respiration represents the most pronounced removal of carbon from the farm/pasture system. As pasture land has a much higher proportional area than forestry, any combined sequestration rate will largely depend on the grassland estimate. The overall sequestration estimates (incorporating both forest and grassland estimates proportional to their relative area) and the resultant footprint are summarized in Table 4. CO_2 emissions due to domestic fuel consumption for the year 2007 were taken from the 2009 national inventory report [33]. The latest population data from the CSO was used to allocate the footprint evenly amongst the population. For the purposes of comparison, Ireland's direct energy footprint for 2007 (as estimated using the standard forestry-based method) is calculated at approximately 2.8 global hectares (gha)/cap.

Table 4. Direct energy footprints using Forest and Grassland sequestration.

	Forest and Farmed Grassland	**Forest and Grassland Net Ecosystem (NEE)**
t C/ha/yr	2.17	3.34
Footprint (ha)	3,862,996	2,515,141
ha/cap	0.91	0.59

As can be seen from Table 4 above, both footprints are significantly less than the energy footprint using global hectares. This suggests that changing the assumptions inherent in the energy footprint calculation may significantly impact the energy (and hence the overall) footprint value.

5. Discussion

5.1. Factors Affecting Forest Sequestration Estimates

The method for calculating carbon sequestration due to annual tree biomass increment is relatively straightforward. A significant point of debate is the inclusion of trees younger than 10 years: many studies [34,35] exclude the sequestration of tree stands below a certain age as it is assumed that they do not have biomass. Because trees below 10 years represent the single largest age category in Ireland, it was considered preferable to include this biomass and potentially overestimate carbon sequestration rather than exclude it and risk a significant underestimation of current annual sequestration. The inclusion of litter fall, deadwood and soil carbon fluxes represents a new addition to ecological footprinting. This is relevant to all footprint studies, as the energy footprint as currently calculated is based on the carbon stored in forest areas through annual tree growth. Because both the carbon accumulated in the litter and dead pools is estimated as a function of annual growth, and includes the compound carbon losses due to decomposition, the results are extremely sensitive to both the values used within the tree biomass calculation as well as the decay rates. One of the difficulties in modeling the decomposition of litter is that the decay of an individual piece of leaf litter or coarse woody debris will change over time. (While most of the following discussion relates to leaf litter, it is equally relevant to the dead pool). The decomposition rate constant refers to the forest floor as a whole but, it will differ from the decay rate of both recently fallen and older litter. This is due to the fact that the most biochemically labile elements will decompose first. Ideally, an annual decay rate (k) of 0.10 yr^{-1} suggests a litter residency time of 10 years. Thomas and Packham [36] report on an early model [37], which suggests that due to the incremental reduction in litter decay rates an individual litter fall may take a period of 5 /k to lose 99% of its original mass. The method of calculating decomposition losses within this paper was compound based but maintained the same decay rate throughout. However, it is difficult to deduce the result of such a change as it has been suggested that while broadleaf and coniferous decay rates converge over time, the changes to k leading up to convergence may vary significantly [38]. There appears to be considerable debate within the literature as to what are the main determinants of leaf litter decomposition. Indeed, the variable—and often contradictory—estimates of forest litter dynamics may negate the value of using decay constants extrapolated from experimental data. In many cases correlation between litter decay rates and external factors may not adequately reflect cause and effect and may mask the impact of potentially more influential factors, such as temperature of precipitation [39].

The estimate for the carbon soil flux represents a conservative estimate: this value was adopted due to the large variance in soil respiration rates [40]. As NEE is defined as the difference between GPP and combined autotrophic and heterotrophic respiration, an increase in soil respiration will decrease the carbon stored in soil. Saiz *et al.* [26] publish soil respiration estimates ranging between 991 and 564 g $C/m^2/yr$ for three different mineral gley sites (with a stand age of 10 to 31 years) and a 47 year old gleyic brown earth. Valentini *et al.* [8] in examining European soils state that respiration, because of its variability in comparison to NEE, is the main determinant of sequestration. Because of these uncertainties, the lower value for the sequestration rate of Irish forest soil, taken from [28], was applied to all stands older than 10 years.

5.2. Methodological and Conceptual Options for Estimating Grassland Sequestration

The inclusion of two alternative grassland sequestration estimates within this study is related to the discussion of single land use and the energy footprint as a theoretical or notional measurement. For that reason, two estimates of grassland sequestration were adopted in Tables 3 and 4, and reflect different choices of study boundaries. The study, which provided the data seen in Table 2, viewed the farm gate as the boundary between the technosphere and the ecosphere. While this is at odds with a strict definition of what constitutes an 'ecosystem', it is based on the perception that an area of productive farm land is not a natural system and its role in the movement of fixed carbon should be acknowledged. In other words, the natural processes within a grazed area cannot be viewed in isolation, and so inclusion of lateral carbon movement from the farm provides a more realistic appraisal of the potential sequestration services provided by agricultural grassland. By contrast, if the boundary is simply placed around the grassland (above and below ground), then NEE estimated by eddy covariance (while removing emissions due to respiration, *etc*.) will be suitable. This is an attempt to only account for the natural process in the soil which adds to the carbon stock. A footprint calculated using a farm-based estimate provides the farmed area needed to sequester a given quantity of carbon, whereas applying an estimate of NEE is intended to represent an evaluation of the sequestration of grassland in a more natural state. Footprint practitioners may well suggest that since farm activities can reduce the natural carbon sequestration rate, a productive grazing area will not function as the optimal grassland sink and therefore should be not be included within ecological footprinting. Therefore, the rationale for adopting NEE is based on the fact that it corresponds to current practice as applied to forest areas. Whereas a farm-based footprint represents an area that may currently offset carbon emissions, a footprint calculated using NEE represents both the additional land area that would be required and also the optimal grassland type for long term carbon sequestration.

The question of whether a farm based sequestration estimate is suitable within ecological footprinting is more complex than simply a discussion of the theoretical nature of the energy footprint would suggest. The application of NEE assumes that the grassland area will not be disturbed, and maintained specifically for the purpose of carbon sequestration. However, it may be argued that farm production and grassland sequestration are closely linked [41]. As grassland sequestration is related to productivity, the estimates of NEE are related to farm practices. For example, the application of fertilizers, both organic and inorganic, may increase grass production and subsequently increase NEE. Given the intensive nature of modern Irish grassland management, it may be the case that an individual estimate of NEE is inextricably linked to the grazing herd and farm management structure regardless of whether it is calculated as part of a farm carbon balance or not. If this is accepted, then some commentators may view the concept of 'natural' or 'unimproved' NEE as unrealistic within an Irish context. However, that does not mean that the lateral movement of carbon should be accepted without question. The discussion of farm products in relation to carbon sequestration is related to the issue of double counting and single land use. Any removal of carbon in farm products is carbon that has been removed from the grassland area and has effectively lost the opportunity to enter that carbon pool. While farm products would already be included in a national footprint (increasing the footprint as they represent a demand placed on bio-capacity), their inclusion within the lateral movement of a farm carbon balance will reduce the sequestration rate and therefore increase the energy footprint. In other words, the production of a single commodity will impact two distinct footprint types, energy and pastureland. This may result in contradictory data, as productive grassland can have a high yield which means less area is required to produce a commodity. (Note: a high yield may result in a higher footprint if measured in global hectares as a greater area of global grassland is necessary to produce the same amount of goods). However, the more productive an area of grassland becomes, the more carbon will leave the farm boundary in the form of agricultural products and the amount of carbon available for inclusion within the soil pool deceases. The issue of double counting may suggest that unaltered NEE is more suitable for inclusion within a compound footprint such as the global footprint network (GFN) national footprint accounts, which already incorporate the footprint of agricultural products [42].

5.3. Uncertainties

To place the above discussion in better context, the main uncertainties are listed in Table 5 below. It should be stated that while sequestration due to tree biomass represents the highest sequestration estimate, it also incurs comparatively lower uncertainly as it incorporates national data as well as data from reliable sources. However the relatively low area of forestry (when compared with pastureland) means this is less relevant to the overall estimates seen in Table 4.

Given the inherent uncertainty in such a methodology, perhaps the grassland estimates generated in this study should be regarded as a case study, representing a specific area. In that regard, the methods shown in this paper should be viewed as a seeking to inform methodological debate as opposed to accurately represent the whole area of Irish grassland.

Table 5. Estimation of main uncertainties.

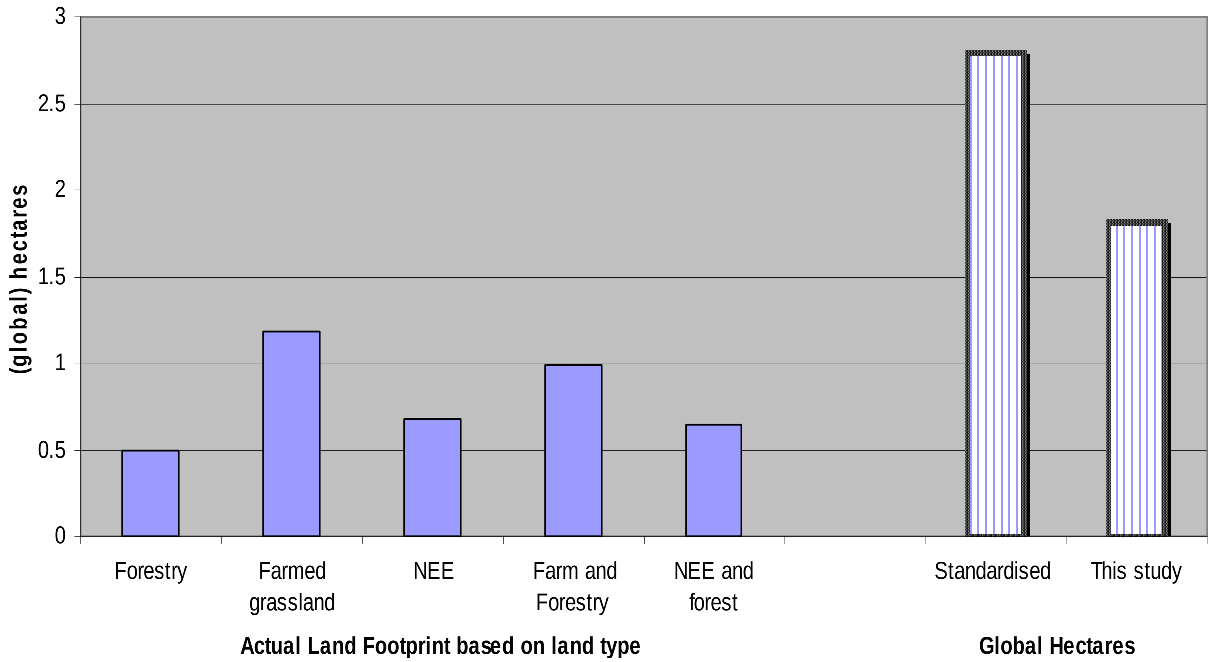
Parameter	**Level of uncertainty**	**Impact**
Decomposition rate	High	Significant for litter and deadpool estimate. However unlikely to greatly impact overall forest sequestration estimation.
Tree mortality	Medium if applied nationally; high if applied to a specific area.	Significant for deadwood estimate.
Forest soil sequestration	High	Unlikely to be significant in comparison to tree biomass estimate.
Animal respiration	Medium, as based on published results, but may not fully reflect outdoor conditions.	Potentially significant for modified NEE and hence overall estimate.
NEE	Medium to high.	Depends on the variance between grassland areas mentioned in [15] and national areas. Potentially significant.

5.4. Estimating Results in Terms of Global Hectares

For purposes of comparison with the existing method, the expanded sequestration footprint was translated into global hectares. While the equivalence factors necessary to translate specific land types at global productivity into global hectares are already published [1], the yield factors necessary to first translate national areas into specific land areas of world average productivity must be estimated. The 2004 *World Academic Footprint Account* [41] estimates that global annual forest increment and grassland yields are 1.84 m^3/ha/yr and 2.23 tonnes of dry matter (t DM)/ha respectively. A value of 10.5 t DM/ha/yr for average grass yields was estimated based on data from [15] and [41], while an average forest increment of 9.9 m^3/ha/yr was estimated based on the BFC yield tables [21] and the national species distribution [17]. These estimates resulted in a yield factor of 4.7 and 5.4 for forestry and pastureland, respectively. The direct energy footprint (in terms of global hectares) was recalculated using an overall sequestration estimate measured in global hectares. This was calculated by allocating the forest sequestration and corrected grassland NEE estimate, yield and equivalence factors to the relative area of both forest and grassland. The overall value was calculated at 1.82 gha/cap, or 65% of the calculated value, when global forestry is translated into global hectares (*i.e.*, reduced by 35%).

As can be seen from Figure 2, calculation of the ecological footprint resulting from domestic fuel consumption, in terms of actual hectares, results in less value than the estimate from current standardized footprint methods. However, as such values represent different units—Irish hectares and global hectares, respectively—they are not strictly comparable. It could be argued that actual footprints, representing a very specific area, have little comparative value on their own and should only be presented in association with a footprint calculated using the standardized method and expressed in global hectares.

Figure 2. Comparison of footprint results: comparing actual and global hectares.



Perhaps more interesting is that translating the forest and NEE based footprint into global hectare equivalents results in a lower value than if the global hectares are calculated normally (using the sequestration estimate of global forestry and the associated equivalence factor.) This may be due to the fact that although Irish grassland and forestry are significantly more productive (resulting in a large yield factor) than the global average, pasture land has a lower equivalence factor than forestry. While Irish forestry and grassland incur a similar cost (*i.e.*, an increase in the footprint) when translated into global forestry and global grassland, grassland has an equivalence factor that is approximately one-third of the equivalent value for forestry. As suggested earlier, there is no guarantee that the same relationship between biomass and litter/deadwood accumulation will be evident on a geographic scale or when translating forest areas into global hectares. Also, the inclusion of soil processes challenges the notion of the energy footprint as an optimal area. It is unlikely that any area devoted to carbon capture will function wholly as a carbon sink; certain areas within a 'carbon footprint' area will occasionally function as a carbon source.

6. Conclusions

The need for ready applicability and comparability within ecological footprint analysis is manifest. However, the need to suitably take local conditions into account is justifiably as important. The expansion of the carbon footprint method to include additional pools and local conditions has been raised as a possible methodological improvement. It has been shown that the inclusion of other carbon pools is possible within a forest based footprint, the practicality of such would depend on data accuracy. The degree to which such methods may be integrated into a standardized ecological footprint method are contingent on the availability of data on a global scale. The inclusion of Ireland's extensive and intensive pasture area reduces the direct energy footprint, but has relevance to the issue of energy land as a notional metric. The inclusion of lateral carbon movement and removals due to grazing reduces the sequestration rate of grassland and may constitute double counting. However, its addition to forestry-based energy conversion factor results in a considerable footprint reduction when translated into global hectares. In summary it is unlikely that a global average is capable of addressing the variability of local (or indeed national) conditions. In other words, the eventual footprint value is largely dependent on individual methodological choices, which reaffirm the need for clear caveats when using national conditions to infer global bio-productive demands and *vice versa*.

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