GCB Bioenergy (2012) **4**, 311–329, doi: 10.1111/j.1757-1707.2011.01143.x

Using a Strategic Environmental Assessment framework to quantify the environmental impact of bioenergy plans

JOHN FINNAN*, DAVID STYLES†[1], JOANNE FITZGERALD*, JOHN CONNOLLY‡ and ALISON DONNELLY§

Crops, Environment and Land Use Programme, Teagasc, Oak Park, Carlow, Ireland, †European Commission Joint Research Centre, Seville, Spain, ‡Biosystems Engineering, University College Dublin, Belfield, Dublin, 14, Ireland, §Centre for the Environment and Department of Botany, School of Natural Sciences, Trinity College, Dublin, 2, Ireland

Abstract

Renewable energy and greenhouse gas (GHG) reduction targets are driving an acceleration in the use of bioenergy resources. The environmental impact of national and regional development plans must be assessed in compliance with the EU Strategic Environmental Assessment (SEA) Directive (2001/42/EC). Here, we quantify the environmental impact of an Irish Government bioenergy plan to replace 30% of peat used in three peat-burning power stations, located within the midlands region, with biomass. Four plan alternatives for supplying biomass to the power plant were considered in this study: (1) importation of palm kernel shell from south-east Asia, (2) importation of olive cake pellets from Spain and (3) growing either willow or (4) *Miscanthus* in the vicinity of the power stations. The impact of each alternative on each of the environmental receptors proposed in the SEA Directive was first quantified before the data were normalized on either an Irish, regional or global scale. Positive environmental impacts were very small compared to the negative environmental impacts for each of the plan alternatives considered. Comparison of normalized indicator values confirmed that the adverse environmental consequences of each plan alternative are concentrated at the location where the biomass is produced. The analysis showed that the adverse environmental consequences of biomass importation are substantially greater than those associated with the use of willow and *Miscanthus* grown on former grassland. The use of olive cake pellets had a greater adverse environmental effect compared to the use of peat whereas replacement of peat with either willow or *Miscanthus* feedstocks led to a substantial reduction in environmental pressure. The proposed assessment framework combines the scope of SEA with the quantitative benefits of life cycle assessment and can be used to evaluate the environmental consequences of bioenergy plans.

Keywords: *Miscanthus*, olive cake, palm kernel, peat, Strategic Environmental Assessment, willow

Received 28 September 2011; revised version received 28 September 2011 and accepted 5 October 2011

Introduction

A desire for reduced dependence on fossil fuels together with growing evidence of the effect of increasing greenhouse gas (GHG) emissions on climate (Solomon *et al*., 2007) is driving interest in renewable forms of energy, including bioenergy. Following on from GHG emission reduction targets established under the Kyoto Protocol of the United Nations Framework Convention on Climate Change (United Nations, 1998), the European Union has committed to a 20% reduction in GHG emissions by 2020 with 20% of energy coming from renewable resources. Renewable energy targets for EU member states are set out in the Renewable Energy Directive (2009/28/EC).

Correspondence: John Finnan, tel. + 00 353 599 170 253, fax + 00 353 599 142 423, e-mail: john.finnan@teagasc.ie

[1]The views expressed are purely those of the author and may not in any circumstances be regarded as stating an official position of the European Commission.

Co-firing biomass with fossil fuels offers a way of increasing renewable energy generation in existing power stations. In the United Kingdom, the Renewable Obligation order (DECC, 2009) places a requirement on UK suppliers of electricity to source an increasing proportion of their fuel from renewable sources. As a consequence, a large number of power stations in the United Kingdom employ biomass co-firing (Woods *et al*., 2006). In Ireland, the recent white paper on energy policy (DCMNR, 2007) includes a target to co-fire 30% biomass in the country's three remaining peat-burning power stations. Demand for biomass for co-firing and other bioenergy uses is set to increase as 2020 approaches and EU countries strive to achieve their bioenergy targets.

The expansion of bioenergy is largely driven by energy security and environmental concerns. It is imperative, therefore, that the most environmentally friendly and sustainable biomass supply options are selected when fossil energy is replaced by bioenergy. The EU Renewable Energy Directive introduced sustainability requirements for the supply of biofuels and bioliquids. Reviews have been conducted to assess the environmental impact of energy crop production (Haughton *et al*., 2009; Rowe *et al*., 2009; Dauber *et al*., 2010). Additionally, Haughton *et al.* (2009) suggested a sustainability framework approach based on indicators to understand the widespread environmental consequences of large-scale energy crop introduction. However, both of these approaches used a limited number of environmental receptors (water, air, biodiversity, etc.) and were nonquantitative. In contrast, Schmidt (2007) used a quantitative approach based on life cycle assessment (LCA) to compare oil production from rape in Denmark to palm oil production in the Far East. LCA can provide quantitative assessments of the effects of processes on environmental receptors such as climate, acidification, eutrophication, etc. However, LCA typically covers a limited range of impacts that can be quantitatively assessed (e.g. resource depletion, impacts such as air and water pollution and climate change attributable to quantifiable emissions, waste generation), and may not integrate results across impact categories, although aggregation procedures have been proposed for certain applications (Brentrup *et al*., 2004).

The Strategic Environmental Assessment (SEA) Directive introduced by the European Union in 2001 (2001/42/EC) proposes a comprehensive range of environmental receptors to define the environment: biodiversity, population, human health, fauna, flora, soil, water, air, climatic factors, material assets, cultural heritage including architectural and archaeological heritage and landscape. The use of such a wide range of receptors ensures a comprehensive environmental assessment. Previously, it was shown that SEA could provide a practical and comprehensive framework for an assessment of the environmental impact of bioenergy plans (Donnelly *et al*., 2011). The objective of this study was to determine if the approach adopted by Donnelly *et al.* (2011) could be further developed to provide an integrated quantitative assessment of bioenergy plans and programmes based on the comprehensive description of the environment provided by SEA complemented with quantitative LCA methodology. The Irish 30% co-firing target was used as a relevant case study for testing the approach of Donnelly *et al.* (2011).

Method

The plan in the case of this study was to supply 30% of the energy input to the three remaining peat-burning power stations in Ireland with biomass. The three power stations are located in the midlands of Ireland at Edenderry in Co. Offaly, Lanesboro in Co. Longford (Lough Ree Power Station) and Shannonbridge in Co. Offaly (West Offaly Power Station).

Four plan alternatives for supplying the biomass feedstock were considered.

1 Palm kernel shell from Indonesia.
2 Olive cake pellets from Spain.
3 *Miscanthus* grown in a 50 km radius of each power station.
4 Willow grown in a 50 km radius of each power station.

The environmental impact of each of the plan alternatives was quantified using a hybrid approach combining the environmental receptors proposed in the European Union SEA Directive (2001/42/EC) with impact assessment based on LCA methodologies where available (Table 1). The environmental impact of continuing to use peat to supply 30% of the power plants feedstock requirements was also quantified as a baseline. The procedure developed to evaluate the environmental impact of each plan alternative may be applied to any plan or programme. The steps in this procedure are defined as follows:

Table 1 Methods used in the study to quantify the impact of the plan on the environmental receptors defined in the SEA Directive (2001/42/EC)

SEA receptors	Common LCA methods	Other methods
Climate	GWP (CO_2 eq)	
Air	Acidification (SO_2 eq), human health (1,4-DCB eq), tropospheric ozone formation (NMVOC eq)	
Water	Eutrophication (PO_4 eq), freshwater ecotoxicity (1,4-DCB eq), water footprint (litres appropriated)	
Biodiversity	Species richness (see Table 3)	
Soil		Waste dumped to landfill
Material assets		Road wear (km tonnes transported by road)
Landscape		Landscape index (see Fig. 2)
Population and human health		Number of direct jobs
Cultural heritage		Areas of national/international importance affected by the plan

SEA, Strategic Environmental Assessment; LCA, life cycle analysis; NMVOC, nonmethane volatile organic compound; 1,4-DCB, 1-4 dichlorobenzene.

1 Distinct phases in the proposed plans are identified.
2 Environmental receptors appropriate to the proposed plan are chosen from the list proposed in the SEA Directive.
3 The effects of the proposed plan on each environmental receptor identified in step two are quantified using relevant indicators.
4 The indicators are normalized against appropriate data. Normalized indicators may either be negative, neutral or positive according to the effect of the proposed plan on an environmental receptor.
5 The normalized indicators relevant to each environmental receptor are averaged to produce a comparable value for each environmental receptor.
6 Receptor values are summed to produce an aggregate indicator summarizing the environmental impact of the plan. Environmental receptors may be weighted equally or differently depending on the objectives of the plan. In the case of this bioenergy plan, each receptor was weighted equally.

The concept is illustrated in Fig. 1 which explains how the advantages of SEA and LCA are combined in this study. In this study, the above procedure was carried out for each of the four plan alternatives described as well as for the default alternative in which the 30% alternative feedstock target continues to be supplied by peat. All of the eight environmental receptors specifically referred to in the SEA Directive were considered applicable to this bioenergy plan (see Table 1). This, however, will not necessarily be the case for all development plans.

Environmental receptors in SEA do not correspond directly with environmental impact categories in LCA. Some environmental receptors are associated with a number of relevant environmental impact categories from LCA, others with one or with poorly developed LCA impact categories (e.g. biodiversity). For example, there are three relevant indicators for air quality (acidification, ecotoxicity and ozone formation), but just one relevant indicator for climate [the Global Warming Potential (GWP) of GHG emissions]. To maintain the comprehensive SEA framework, and use all relevant LCA indicators in the assessment procedure without giving additional weight to environmental receptors having a greater number of relevant available indicators, it was decided to average multiple normalized LCA indicator values relevant to each environmental receptor. This assumes equal weighting of LCA impact categories within each environmental receptor, and lower weighting of LCA impact categories where multiple categories are relevant to a single receptor, in accordance with maintaining the SEA structure.

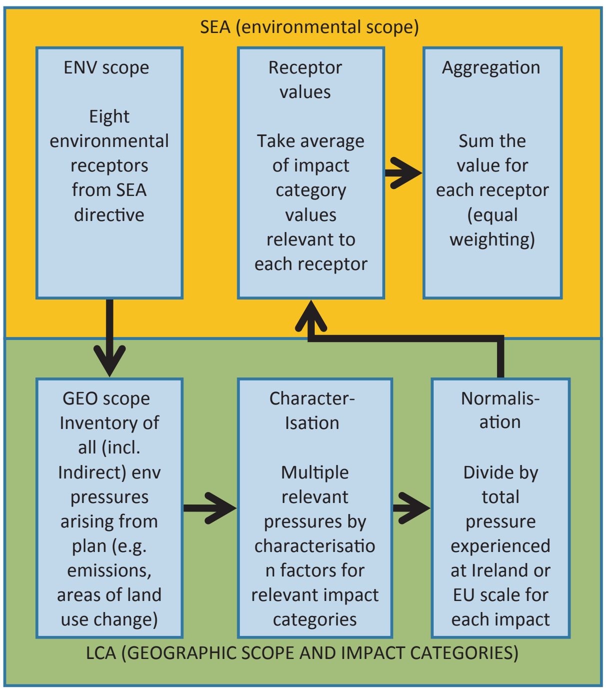

Fig. 1 Concept of the study: life cycle assessment is expanded by incorporating the range of environmental receptors utilized in Strategic Environmental Assessment.

Systems boundary

An LCA approach was taken to define geographical and process systems boundaries for each of the plan alternatives assessed in this study (Fig. 1). Boundaries are described in Table 2. Within the systems boundaries, the environmental impacts of each alternative were quantified and compared. All impacts of land transformation and subsequent crop production were quantified including the GHG budgets associated with fertilizer and pesticide manufacture. Impacts of the plan were considered from a life-cycle perspective, from cradle to grave. In the case of palm and olive fruits, the environmental impacts of crop transport to the mill and of crop processing were assessed. Similarly, the environmental impacts of feedstock transportation to the power station and of its subsequent combustion were assessed. The environmental impacts of palm kernel and olive cake up to the farm gate were allocated using a energy-balance approach. For peat, the environmental consequences of bog drainage, vegetation removal, peat harvesting and transport were included in the systems boundary and assessed. The environmental impact of power station construction and of emissions to water from power station operation were not included in this analysis as these were assumed to be common to all plan alternatives under consideration.

All environmental impacts were calculated over a period of 1 year with the exception of the effect of land transformation on biodiversity, which were defined by Schmidt (2008) as an effect over a period of renaturalization. The functional unit is defined as 30% of the energy output of the three peat power stations over 1 year (1000 GWh).

Plan alternatives

The four alternatives for supplying biomass for the co-firing plan are described as follows.

Olive cake pellets

Olive farming is a significant land use in Mediterranean countries, the principal olive producing country is Spain followed

Table 2 Summary of baseline and biomass co-firing plan alternatives in the three peat power stations, and associated system boundaries, scope and allocation methods considered in the standard analysis

Plan alternative	Feedstock and system boundaries	Allocation and scope	Functional unit
Baseline 100% peat firing	Drainage and cutting of peat bogs; transport and preparation of peat; power station operation; ash disposal Excluded: power station construction	30% of all impacts (as baseline for 30% co-firing)	30% peat power station energy output 1000 GWh electricity generation per year
Olive cake pellet co-firing 30%	Land conversion to olive cultivation in Andalucia, Spain (from cropland, grassland, scrubland and forest land) Olive cultivation on 181 245 ha Olive mill operations Olive cake processing and pelleting Olive cake pellet transport to peat power stations in Ireland Power station operation Ash disposal	Based on oil and olive cake energy balance, 38% of all land transformation and cultivation impacts, and olive mill activities associated with oil extraction 100% olive cake processing, pelleting and transport impact 100% of impact calculated for combustion of olive cake pellets in the peat power station, including ash disposal	
Oil palm kernel shell co-firing 30%	Land conversion to palm oil cultivation in Indonesia, from disturbed agricultural land Palm oil on 291 666 ha Palm oil mill operations Palm kernel shell transport to peat power stations in Ireland Power station operation Ash disposal	Based on kernel shell and palm oil energy balance, 11% of all land transformation and cultivation impacts, and mill activities associated with oil extraction 100% palm kernel shell transport impact 100% of impact calculated for combustion of palm kernel shell in the peat power station, including ash disposal	
Miscanthus co-firing 30%	Land conversion to *Miscanthus* cultivation from grassland in Ireland; *Miscanthus* cultivation on 45 000 ha within 50 km of peat power stations; transport of harvested *Miscanthus* to peat power stations; power station operation; ash disposal	100% of impacts over 21 year plantation lifetime, divided by 21	
Willow co-firing 30%	Land conversion to willow cultivation from grassland in Ireland Willow cultivation on 45 000 ha within 50 km of peat power stations Transport of harvested willow to peat power stations Power station operation Ash disposal	100% of impacts over 21 year plantation lifetime, divided by 21	

by Italy, Greece and Portugal (Beaufoy, 2000). Oil is extracted from olives leaving a solid byproduct/residue and, depending on the process, a liquid waste. The solid waste (olive cake) can be pelleted as a biofuel.

The focus of this study was on olive cake pellets produced in Spain, the largest producer of olive oil in Europe (Beaufoy, 2000; MORE, 2008). The Spanish olive oil industry is concentrated in the province of Andalusia in the southern part of the Iberian Peninsula. Traditional olive plantations in Spain used a low tree density and were often planted on hills and mountainous areas, and frequently associated with soil erosion (Beaufoy, 2000). However, recent decades have seen a switch to intensive plantations typically planted on rolling and flat plains. These plantations are characterized by a tree density of between 200 and 400 trees per hectare consisting of short stem varieties, which may be re-planted every 25–30 years (Beaufoy, 2000) and may yield up to 10 t of olives per hectare. We assumed that olive production in Spain was concentrated on these plantations. Such plantations have been established from land previously under arable crops, grassland, scrub and forest (Beaufoy, 2000). For this study, one quarter of the land converted to growing olives was assumed to have been in each of these previous uses. Values for fertilizer use, chemical use and water use were taken from Beaufoy (2000), whereas the value for diesel use during cultivation was taken from Avraamaides & Fatta (2008). An average yield of 9 t of olives per hectare was assumed (Beaufoy, 2000).

After harvest, olives are taken to an olive mill. Most olive mills in Spain utilize a two-phase extraction system, which initially produces a moist waste material which is subsequently dried as part of the process to produce a solid waste material with a moisture content of ca. 10%. This material is suitable for pelleting. Small amounts of liquid waste are produced in the process (MORE, 2008).

La Cal (2010) reported an annual production of 530 000 t of olive oil in the province of Jaen compared to 650 000 t of exhausted pomace (olive cake).

The electricity consumption of the processing plant (0.23 kWh per litre olive oil) was taken from Avraamaides & Fatta (2008). Energy requirements of the pelleting process were taken from the pellet handbook (Obernberger & Thek, 2010), 18.7 kWh t^{-1} for the grinding process, 38 kWh t^{-1} for the pelleting process and 1.5 kWh t^{-1} for the cooling process.

This plan alternative is to supply the biomass required for co-firing from olive cake produced from the olive oil processing industry in Spain. Considerable quantities of olive waste (olive cake) are produced during the oil manufacturing process (MORE, 2008) which are being sold on the world market, typically as cake, pellets or expeller (Woods *et al*., 2006). It was assumed that olive cake pellets would be used in the plan. After manufacture, it was assumed that pellets were transported an average distance of 150 km to a port in Spain before being shipped to a port in Ireland (2141 km) and transported by road from a port in Ireland to one of the three peat-burning power stations (100 km).

Palm kernel shells

Oil palms grow to ca. 10 m in height and have a productive life-span of 25 years. The fruits are harvested as fresh fruit bunches (FFB) which are brought to an oil mill where the oil is extracted from the fruit and, in a secondary step, often from the kernel of the fruit as well (Schmidt, 2007). The initial oil pressing stage produces shell fragments and fibre as by products. The shells and fibre are often used as a fuel to provide heat and electricity for the oil mill although sufficient quantities are produced in Malaysia to more than double levels of co-firing in the United Kingdom (Woods *et al*., 2006).

Oil palm production is concentrated in Malaysia and Indonesia (FAOSTAT, 2009). Schmidt (2007) conducted a detailed assessment of palm oil production in Indonesia and Malaysia and reported that oil palm plantations were almost always established on disturbed agricultural land rather than by replacing virgin forests. In his study, Schmidt assumed that 50% of land used to plant palm trees was previously grassland and 50% of land was degraded forest, these land transformation figures were used in this study. External inputs to the system include fertilizer (105 kg N; 31 kg P and 170 kg K ha^{-1} yr^{-1}), pesticide (2.7 kg active ingredient ha^{-1} yr^{-1}) and energy (2118 MJ ha^{-1} yr^{-1}). An average yield of 18.74 t of FFB per hectare was used (Schmidt, 2007). Data on oil mill electricity use, water usage and emissions to air were taken from Schmidt (2007).

This plan alternative involves the use of palm kernel shells to supply the biomass required by the 30% co-firing plan. The shell produced at the oil mill was transported an average distance of 200 km to a port in Indonesia before being transported to a port in Ireland (15 500 km) and then transported an average distance of 100 km to be co-fired in one of the three peat-burning power stations.

Miscanthus

There are ca. 2500 ha of *Miscanthus* in Ireland at present (McDonagh, 2010). However, relatively little is grown in the immediate vicinity of the three peat-burning power stations at present. Consequently, the *Miscanthus* required for co-firing in this plan alternative would have to be established and grown in the vicinity of the power stations. It is assumed that the *Miscanthus* will be established in former grassland as grassland occupies ca. 90% of the land area in Ireland (O'Mara, 2008) with the majority of cereal production concentrated in the East and the South. It was assumed that *Miscanthus* for co-firing would be sown within a radius of 50 km of each of the three peat-burning power stations. The first stage of ground preparation includes herbicide application followed by subsoiling and ploughing. Rhizomes are planted in the spring following rotavation, ridging and pick-up of 3 year old *Miscanthus* rhizomes where 1 ha supplied rhizomes are used to plant 10 ha at 20 000 rhizomes ha^{-1} at a total energy intensity of 4000 MJ ha^{-1} (Bullard & Metcalf, 2001). It was assumed that lime was applied four times during the 21-year life cycle. Average yields of 10 t of dry matter per hectare per annum were assumed over the lifetime of the plantation. Herbicide application was assumed to consist of two preplanting applications, one application in each of the first 3 years and thereafter every 2 years, two herbicide applications were assumed to be necessary to remove the crop. It was assumed that no fertilizer was used in the first 2 years and in the last year. Maximum fertilization rates of 100 kg N ha^{-1} were suggested by Lewandowski *et al*. (1995). N requirements for *Miscanthus* were defined by Plunkett (2010) to vary between 30 and 100 kg N ha^{-1} depending on soil nutrient status. Clifton-Brown *et al*. (2007) and Riche *et al*. (2008) reported that the yield of *Miscanthus* which received no fertilization declined after a period of time. For this study, we assumed that nitrogen fertilization was necessary to replace crop offtakes and that nitrogen fertilization rates ranged from 50 to 100 kg N ha^{-1} with a midpoint of 75 kg N ha^{-1}.

At harvest, it was assumed that *Miscanthus* was mowed and then baled before being transported an average distance of 25 km to be co-fired in one of the three peat-burning power stations.

Short rotation coppice willow

There are ca. 500 ha of willow in Ireland at present (McDonagh, 2010). However, relatively little is grown in the immediate vicinity of the three peat-burning power stations at present.

Consequently, the willow required for co-firing in this plan alternative would have to be established and grown in the vicinity of the power stations. It is assumed that the willow would be established in former grassland as grassland occupies ca. 90% of the land area in Ireland (O'Mara, 2008) with the majority of cereal production concentrated in the East and the South of the country. It was assumed that willow for co-firing would be sown within a radius of 50 km of each of the three peat-burning power stations. It was also assumed that the plantation life will be 21 years, consisting of cutback after Year 1 followed by 10 harvests at 2-year intervals. Planting is preceded by two herbicide applications, subsoiling, ploughing and tilling. Coppicing (cut-back) in Year 1 and each subsequent harvest with the exception of the last harvest is followed by a herbicide application and by fertilization. The last harvest is succeeded by two herbicide applications to kill the crop and ploughing to remove the crop. Yields from the first cropping cycle can be expected to be lower than subsequent cycles because of incomplete site capture before yields reach a plateau with normal variation due to prevailing weather conditions (Dawson, 2007). Average yields of 10 t of dry matter per hectare per annum were assumed over the lifetime of the plantation. Fertilization rates up to 120–150 kg nitrogen, 15–40 kg phosphorus and 40 kg potassium per hectare per year have been suggested by Dawson (2007). Plunkett (2010) suggested nutrient application rates of 40–130 kg N ha^{-1} yr^{-1}, 0–34 kg P ha^{-1} yr^{-1} and 0–155 kg K ha^{-1} yr^{-1} depending on the nutrient levels in the soil. For this study, it was assumed that fertilization of willow is necessary to replace crop offtakes and that nitrogen fertilization rates ranged from 50 to 130 kg N ha^{-1} yr^{-1} with a mid-point of 90 kg N ha^{-1} yr^{-1}. There are two principal methods of harvesting willow; the crop can be cut and chipped in one operation after which the chips need to be dried immediately. Alternatively, the crop can be cut as whole stems and left to season before chipping (Dawson, 2007). The latter plan alternative was assumed in the calculation of the energy required to grow the crop as it was assumed that natural drying would be used.

Peat

The use of peat as a feedstock for energy production is described as follows according to Connolly & Rooney (1997).

Peatland is prepared for milled peat production by first removing the layer of vegetation growing on the surface of the bog before drainage ditches or canals are dug into the virgin peatland. The surface of the bog is then levelled to permit machine access and sloped in the direction of the drainage ditches to promote the drainage of surface water (Connolly & Rooney, 1997). For this study, it was assumed that peat was harvested from peatland that was in it natural state before vegetation removal, drainage and levelling had been carried out. Peat is harvested during summer, an operation which consists of milling, harrowing, ridging and stockpiling. Peat is transported from the bogs to the power stations for the most part by means of a system of narrow gauge railways, Connolly & Rooney (1997) assumed that 90% of the peat is transported in this way with the remainder of the peat being transported by road, these assumptions were used in this study.

The preparation and use of virgin peatland has a number of negative environmental consequences. The most serious environmental consequences are associated with the preparation of the peatland for harvesting as the visual impact of the landscape is changed and an increasingly rare habitat is removed. On milled peatlands, the capacity to sequester carbon is removed after the vegetation has been removed and the peatland will not be able to sequester any carbon during the period over which peat is harvested (Waddington *et al.*, 2002). CO_2 emissions increase as a result of increased C oxidation, whereas emissions of CH_4 and N_2O are reduced (Styles & Jones, 2007). Thus, carbon sequestration is reduced to zero and the peatland is converted from a carbon sink to a source of carbon. Other negative environmental consequences arise from wind-blown dust during harvest time, the siltation of rivers from particles washed from the peatland and transport emissions (Connolly & Rooney, 1997).

Phases of the bioenergy plan

A previous assessment of the environmental impact of plans in which crops were introduced (Donnelly *et al.*, 2011) showed that the environmental impact of such bioenergy plans could be divided into two phases, a land transformation phase and a mature phase. Accordingly, the environmental assessment of the four crops in this study (oil palm, olives, willow and *Miscanthus*) was divided into these two phases. During the mature phase of the bioenergy plan, the impact of each plan alternative on all environmental receptors was quantified. However, a limited number of receptors were assessed during the land transformation phase of the bioenergy plan (water, biodiversity climatic factors) as only these receptors were considered relevant to this part of the plan.

Environmental receptors

The impacts of the bioenergy plan on all of the environmental receptors proposed in the SEA Directive (2001/42/EC) were quantified. Several aspects of any of the environmental receptors could be influenced by the plan. For example, emissions to air could be as oxides of nitrogen (NO_x), sulphur dioxide (SO_2), carbon monoxide (CO), particulate matter (PM), ammonia (NH_3) or as organic substances such as pesticides. Water quality could be affected by emissions of nitrogen (N) and phosphorus (P) containing substances and organic compounds to water bodies. Additionally, water use can be affected by a plan.

Biodiversity

The biodiversity indicator was based on the work of Schmidt (2008) and Kollner (2003) who characterized the species richness of a range of global land uses. In our study, changes in species richness per hectare according to Schmidt's values were multiplied by land usage in the bioenergy plan before being normalized.

Species richness values for palm plantations were obtained from Schmidt (2008), whereas those for *Miscanthus*, willow and olive groves were obtained from Kollner (2003). Kollner gives a value for the species richness of *Miscanthus* but not for willow. Haughton *et al.* (2009) used butterflies as an indicator of biodiversity and found differences in butterflies between *Miscanthus* and willow plantations. However, Dauber *et al.* (2010) in a review on the impact of biomass crop cultivation on temperate biodiversity found that there were few studies which compared both crops. In view of the fact that there is little data published on the differences in biodiversity between willow and *Miscanthus* and the fact that no studies on this subject have been published in Ireland, we assumed the value for the biodiversity of willow is identical to that of *Miscanthus*. The species richness of olive groves was assumed to be that of Kollner's agri-high scenario. Weighted species richness for each crop was calculated by multiplying the species richness figure by an ecosystem vulnerability factor unique to each country. This factor was calculated according to Schmidt (2008). Occupation and transformation factors for each plan alternative were calculated according to Schmidt (2008). Occupation effects were calculated by multiplying the occupation factor of the crop in question by the land area required by that particular plan alternative and normalized against the agricultural land area of Ireland or EU-27 occupied by forest. Transformation effects were calculated from the difference between the transformation factors of the land use(s) being replaced and that of the new crop multiplied by the land area required for that particular plan alternative. Transformation effects were normalized against the transformation effect of converting the agricultural land areas of Ireland or the EU-27 area to nature, forest. Characterization factors for biodiversity are given in Table 3.

Water

The effects of the plan on water quality and water quantity were considered separately. Two environmental impact categories were used to quantify the effect of the plan on water quality; eutrophication potential and freshwater ecotoxicity (see Table 4). Emissions to water appropriate to each impact category were multiplied by characterization factors to generate an environmental indicator for each impact category which was subsequently normalized against national and regional scales. The total effect of the plan on this environmental receptor (water) was calculated by averaging the normalized values of the two impact categories. The characterization factors used in this study are given in Table 4.

Emissions of N and P containing compounds to water bodies were calculated as follows.

Land transformation phase. Nitrogen released as a consequence of land transformation was estimated based on carbon

Table 3 Characterization factors for biodiversity given in units of weighted species richness on a standard area of 100 m^2

Ecosystem	Country	Species richness (100 m^2)	z/LI	Weighted species richness (100 m^2)	Occupation factor (100 m^2 yr^{-1})	Renaturalization time (per year)	Transformation from (100 m^2)	Transformation to (100 m^2)
Grassland intensive	Ireland	17	0.47	8	14.6	10	40	−40
Miscanthus	Ireland	15	0.47	7.05	15.5	10	35.2	−35.2
Willow	Ireland	15	0.47	7.05	15.5	10	35.2	−35.2
Nature forest	Ireland	48	0.47	22.6	0	500	5650	−5650
Peat bog	Ireland	19	0.47	8.9	0	500	2225	−2225
Oil palm	Indonesia	30	0.41	12	28	25	150	−150
Grassland	Indonesia	12	0.41	5	35	5	12.5	−12.5
Forest managed extensive	Indonesia	73	0.41	30	10	36	540	−540
Nature forest	Indonesia	98	0.41	40	0	355	7100	−7100
Cereals	Spain	10	0.46	4.6	17.5	1	2.3	−2.3
Grassland	Spain	17	0.46	7.82	14.3	10	391	−391
Forest managed	Spain	24	0.46	11.0	11.1	50	275	−275
Scrub	Spain	18	0.46	8.3	13.8	500	2075	−2075
Nature forest	Spain	48	0.46	22.1	0	500	5525	−5525
Olive groves	Spain	13	0.46	6	16.1	25	75	−75

z/LI: ecosystem vulnerability factor; occupation factor: difference in species richness between a land use and the reference land use; transformation to/transformation from: transformation impacts after converting land from one land use to another land use.

Table 4 Characterization factors applied to emissions to air and water

Impact category	Emission	GWP	Acidification potential	Human ecotoxicity	Tropospheric ozone formation potential	Eutrophication potential	Freshwater ecotoxicity
Source		Forster *et al.* (2007)	Guinee *et al.* (2007)	Guinee *et al.* (2007)	de Leeuw (2002)	Guinee *et al.* (2007)	Guinee *et al.* (2007)
Unit		CO_2 eq	SO_2 eq	1,4 dichlorobenzene eq	NMVOC eq	P eq	1,4 dichlorobenzene eq
Air	CO_2	1					
	N_2O	298					
	CH_4	25			0.014		
	SO_x		1.2	0.096			
	NO_x		0.5	1.2	1.22	0.043	
	CO				0.11		
	PM_{10}			0.82			
	NH_3		1.6			0.114	
	NMVOC			11.4	1		
	Herbicides			0.00065			
Water	Total N					0.137	
	Total P					1	
	Herbicides						2.7*

NMVOC, nonmethane volatile organic compound.
*Characterization factor of MCPA used (2-methyl-4-chlorophenoxyacetic acid).

loss using a C : N ratio of 15 : 1 (IPCC, 2003) and a distribution of 1.25% N_2O-N, 3.75% N_2-N and 95% NO_3-N (Schmidt, 2007). The quantity of this NO_3-N reaching water was calculated according to the procedure of Brentrup *et al.* (2000) who presented a calculation based on emissions of NH_3, N_2 and N_2O, plant uptake and water exchange frequency. It was assumed that no nitrate was released when carbon loss was negative (gain in soil carbon) or neutral. Calculation of the amount of phosphorus released as a result of soil transformation was based on carbon loss using a C : N : P ratio of 186 : 13 : 1 (Cleveland & Liptzin, 2007). The proportion of released phosphorus emitted to surface water was calculated using a fate factor of 3% (Huijbregts & Seppala, 2001) that reflects the relatively low mobility of phosphorous from soil.

Mature phase. Following nutrient application, emissions of ammonia and nitrous oxide to air and emissions of nitrate to water were calculated according to Brentrup *et al.* (2000). Emissions of phosphorus to water following nutrient application were calculated according to Huijbregts & Seppala (2001) using a fate factor of 3%.

Run-off water from peatlands is greatly increased after drainage and during maintenance of a dry peat surface, Connolly & Rooney (1997) stated that 50 m^3 of sludge per hectare is lost is this way. Most of the silt is trapped in siltation ponds (Bord na Mona, 2009). The quantities of N and P reaching rivers and other water bodies from this source were calculated based on N and P concentrations in the drainage water reported by Bord na Mona (2009) and the statistic of 90% of the sludge being captured in the silt ponds (Connolly & Rooney, 1997).

Water use for field crops was only considered when irrigation water was applied to crops. Of the crops considered, the only crop receiving irrigation water was olive, the use of water by olive was taken from Beaufoy (2000). Additionally, water use during processing operations was quantified (Schmidt, 2007; MORE, 2008).

Air

Emissions of pollutants to air were quantified during field operations, transport of feedstock to the power station and transport of crops for processing when relevant as well as from feedstock combustion in the power stations. The relevant pollutants were considered to be sulphur dioxide (SO_2), oxides of nitrogen (NO_x), carbon monoxide (CO), particulate matter <10 μm in diameter (PM_{10}), ammonia (NH_3), methane (CH_4) and NMVOC. Emissions of ammonia to air were calculated according to Brentrup *et al.* (2000). Emission factors from tractors engaged in field operations were taken from Audsley *et al.* (1997) (particulate matter, CO, NO_x, SO_2) and from Lindgren (2004). Emission factors for transport (particulate matter, CO, SO_2, NO_x) were taken from the European Environment Agency (TERM 28) (EEA, 2010c) with the exception of sulphur dioxide emissions which were taken from the NTM model (NTM, 2009). Emissions of particulate matter were converted to PM_{10} using the ratio of PM_{10} to total suspended particles reported in the European Monitoring and Evaluation Programme (EMEP) database (EMEP, 2011). Emissions to air from palm oil mills were taken from Schmidt (2007). Emissions to air during the generation of the electricity used to provide power to Spanish olive mills were taken from Dones *et al.* (2007).

The primary emissions to air for the three peat-burning power stations are sulphur dioxide, nitrogen dioxide and ash (Edenderry Power, 2009; ESB, 2009a,b). Fly ash emissions are collected by electrostatic precipitation considerably reducing particulate emissions to the atmosphere. The percentage of sulphur and nitrogen released to the atmosphere from each power station was calculated based on emissions data and the percentage of nitrogen and sulphur in the peat fuel (Shier, 2009). NO_x emissions from the plant were assumed to arise entirely from fuel nitrogen as the fluidized bed technology used in the plants operates at temperatures below those required to generate thermal NO_x (Van Loo & Koppejan, 2008; Shier, 2011). These percentages were used to calculate the quantities of nitrogen and sulphur released to atmosphere when the new fuels were introduced to the power plants. N, S and ash content of palm kernel, olive cake and willow were obtained from the Phyllis database (ECN, 2011), whereas the N, S and ash composition of *Miscanthus* were obtained from the *Miscanthus* handbook (Jones & Walsh, 2001). It was assumed that all ash in the fuel would be captured either as fly ash by efficient electrostatic precipitators or as bottom ash.

Emissions to air during peat harvesting and transport were taken from Connolly & Rooney (1997), emissions to air during the combustion of peat were taken from the annual environment reports submitted by each of the three peat-burning power stations to the Environmental Protection Agency in fulfilment of their licencing requirements (Edenderry Power, 2009; ESB, 2009a,b). Dust storms occur on peatlands during dry summer periods (Bord na Mona, 2009). Such events affect residential areas adjacent to the peatlands. The quantity of particulate matter emitted to air as a result of such events was calculated based on the assumption that such events only occur during dry summers (1 year in five) and that 0.1% of peat dry matter is lost to the air during such summers.

Three environmental impact categories were used to quantify the effect of emissions to air on the environment; acidification potential, tropospheric ozone formation potential and human ecotoxicity potential (Table 1). Emissions to air appropriate to each impact category were multiplied by characterization factors to generate an environmental indicator for each impact category which was subsequently normalized against national and regional scales. The total effect of the plan on this environmental receptor (air) was calculated by averaging the normalized values of the three impact categories. The characterization factors used in this study are given in Table 4.

Soil

Soil was affected by the plan when ash from the power station was disposed of by landfilling. Ash collected from the three peat-burning power stations is landfilled at present (Edenderry Power, 2009; ESB, 2009a,b).

Climatic factors

Land transformation phase. Soil carbon lost or gained as a result of land transformation was quantified using a Tier 1 approach (IPCC, 2003). Land transformed into oil palm was assumed to come from grassland (50%) and degraded/secondary forest (50%) (Schmidt, 2007). Land transformed into olive production was assumed to have come equally from a mixture of four different uses (arable, grassland, scrub and forest) (Beaufoy, 2000). Managed grassland was assumed to have been transformed into willow and *Miscanthus*. Default stock factors provided by IPCC were used to calculate soil C content of the previous land use and of the new crop (e.g. palm, olives, willow, *Miscanthus*). The difference was the total stock change which occurred over time. The IPCC use a 20 year period for calculating annual stock changes, this time period was used in this study. Nitrous oxide emissions from soils as a result of the transformation process was calculated according to IPCC (2003) assuming a soil C : N ratio of 15 : 1 and an emission factor of 0.0125 kg N_2O-N kg N^{-1} (nitrous oxide nitrogen per kilogram nitrogen). Negative and neutral soil carbon balances were assumed to have no accompanying N_2O emissions while losses of soil carbon were assumed to be accompanied by emissions of nitrous oxide. Peat was assumed to be extracted from virgin peatland following drainage and vegetation removal as described by Connolly & Rooney (1997). After drainage, there is a reduction in emissions of methane and nitrous oxide but an increase in CO_2 emissions from carbon oxidation. The difference in GHG emissions between the drained peatland and the peatland in its natural state was calculated by Styles & Jones (2007) to be 2908 kg CO_2 eq per hectare per annum, this figure was used in this study.

Mature phase

To calculate energy use and GHG emissions for the agricultural phase of each crop, it was first of all necessary to construct a model of an average farm representing each crop following the example of Styles & Jones (2007). All relevant inputs to the system and associated processes (e.g. fertilizer manufacture) were then summed and converted into a final GWP value expressed as kg CO_2 eq considered over a 100 year timescale, according to Forster *et al.* (2007) (CO_2 = 1, CH_4 = 25, N_2O = 298). Emissions of nitrous oxide to air from fertilizer application were considered separately and calculated according to Brentrup *et al.* (2000). GHG production from electricity production in Spain used during olive fruit processing and subsequent pelleting (0.48 kg CO_2 eq kWh^{-1}) was obtained from Dones *et al.* (2007). Power and heat used during palm fruit processes is obtained from the combustion of biomass feedstocks (Schmidt, 2007). GHG emissions from oil and diesel use were calculated according to Flessa *et al.* (2002) and included indirect emissions. Emissions of CO_2 from transport (ship, truck) were calculated using European Environment Agency emissions factors (TERM 28) (EEA, 2010c).

CO_2 emissions resulting from peat harvesting and transport were taken from Connolly & Rooney (1997). GHG emissions for peat combustion were taken from the annual environmental reports of the power stations (Edenderry Power, 2009; ESB, 2009a,b). In contrast, GHG emissions during the combustion of biomass feedstocks were assumed to be zero.

Material assets

In the case of this bioenergy plan, material assets were considered to be road infrastructure. The number of kilometre tonnes for each of the plan alternatives were calculated and normalized against Irish and EU-27 statistics.

Landscape

The landscape indicator in this study is a weighted value of the land area used by the plan which is subsequently normalized against a land area relevant to the scale of normalization. In the case of bioenergy plans, it is considered that the plan affects the landscape by the introduction of new or additional crops which alter the visual appearance of the landscape and which provide more diverse habitats for flora and fauna, that is, an increase in ecological services. It is assumed that new crops will benefit the visual appearance and the ecological services of the landscape only if they are sown in a dispersed pattern. Furthermore, it is assumed that any positive effects will be reversed after a certain percentage of new crops have been introduced into the landscape even if sown in a dispersed manner. For this study, the visual effect of new crop introductions on the landscape only are considered and an index for quantifying this effect was constructed. Positive effects are assumed to occur from the dispersed introduction of new crops up to a maximum of 10% after which the positive effect declines to reach zero at 30% of land cover (Fig. 2). In order for the index to be calculated, the area of new crops to be introduced by the plan needs to be added to the existing area of the crops in the area affected by the plan.

Cultural heritage

Cultural heritage is defined in this study as sites of archaeological and architectural importance in addition to areas of the

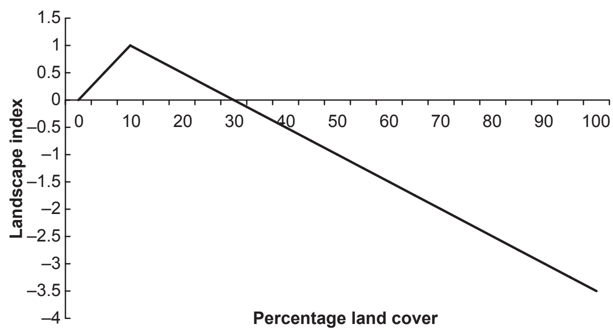

Fig. 2 Landscape index. This index quantifies the effect of percentage crop cover on the visual appearance of the landscape.

country that are unique and precious in terms of national and international conservation importance. Development plans may enhance cultural heritage although typically only if there is a specific provision for the enhancement of cultural heritage in the plan. Typically, it is considered that most development plans will either have a negative or neutral effect on cultural heritage. For this study, the (negative) effects of a particular plan on cultural heritage are considered to be proportional to the level of development in a plan. Development in this context includes new buildings, new roads but also the cultivation or development of previously unused land.

For this bioenergy plan, the only plan alternatives thought to have any effect on cultural heritage as defined above were those with an effect on peatland, a natural habitat and an increasing rare resource. Areas of peatland affected were normalized against the areas of virgin peatland remaining in Ireland and the EU-27 area. This environmental receptor was relevant to the peat alternative as well as for the palm kernel shell alternative where 4% of new palm oil plantations are planted on peat soil (Schmidt, 2007).

Population and human health

Employment was used as a proxy for this environmental receptor. The approach of Thornley *et al.* (2008) was followed although induced employment was not calculated. Employment in the olive industry in Spain was taken from a working paper on the olive oil and table olives sector (EU, 2005), whereas employment in the oil palm industry in south-east Asia was taken from MPOB (2010). Employment generated in Ireland within the agricultural sector from the cultivation and growing of willow and *Miscanthus* was calculated based on diesel usage using a factor of 14 kg diesel per hour (Lindgren, 2004).

Road transport employment was calculated based on the assumptions that drivers worked 8 hour days for 240 days a year transporting 20 t of biomass per load at an average speed of 80 km h^{-1} with 0.5 h for both loading and unloading. Sea transport employment was calculated based on the assumption that the biomass was carried by bulk carriers with a deadweight of 40 000 t (cargo capacity, 34 000 t; Hamelinck *et al.*, 2005) and a crew of 30 travelling at an average speed of 18 km (nautical miles per hour). Employment provided by pellet mills in Spain was taken from Obernberger & Thek (2010) while it was assumed the two additional personnel would be needed at each of the peat power stations to supervise biomass unloading. Employment during peatland preparation, harvesting and transport was taken from Connolly & Rooney (1997).

Allocation

A percentage of the environmental impact of olive production and processing and of palm fruit production and processing was allocated to olive cake and to palm kernel shells. A sensitivity analysis was conducted on the method of allocation in which environmental effects were allocated according to economic value, energy content as well as mass as was done by Luo *et al.* (2009).

Palm kernel shell

Economic allocation for palm kernel shells was calculated based on a price of 1140 US dollars per metric tonne of palm oil (Index Mundi, 2011a) and a price of 74 US dollars per metric tonne of palm kernel shell (BTE, 2011). Economic allocation resulted in 1.93% of the environmental impact of the production and processing of palm fruit being attributed to palm kernel shell.

The energy content of the products of palm fruit processing were obtained from the Phyllis database (ECN, 2011). Allocation according to energy content resulted in 11.14% of the environmental impact of the production and processing of palm fruit being attributed to palm kernel shell.

Mass balance for palm kernel shell was calculated based on products from the oil mill stage of the processing chain using figures from Schmidt (2007). Accordingly, 16.5% of the environmental impact of the palm oil production chain was attributed to the production of palm kernel shell. In total, 291 666 ha of palm oil plantation would be required for the bioenergy plan based on an average yield of 18.74 t ha^{-1} of fresh fruit (Schmidt, 2007) bunches and a requirement of 384 999 t of palm kernel shell for the plan.

Olive cake

Economic allocation for olive cake pellets was calculated based on a price of €2239 per metric tonne of olive oil (Index Mundi, 2011b) and a price of €125 per metric tonne of olive cake pellets. The latter price is based on an average market price of €95 per tonne dried olive cake (Arkady, 2011; Store Energy Renewables, 2011) plus a price for pellet production of €30 per tonne (Obernberger & Thek, 2010). Economic allocation resulted in 2.23% of the environmental impact of the production and processing of palm fruit being attributed to palm kernel shell.

The energy content of the products of olive fruit processing was obtained from the Phyllis database (ECN, 2011). Allocation according to energy content resulted in 38.3% of the environmental impact of the production and processing of palm fruit being attributed to olive cake.

Mass balance for olive cake was calculated based on the two products from the olive mill (olive oil and exhausted pomace). From La Cal (2010), 530 000 t of olive oil and 650 000 t of exhausted pomace are produced from oil mills in the Spanish province of Jaen. Accordingly, 55.2% of the environmental impact of the olive chain was attributed to the production of olive cake. In total, 181 245 ha of olive would be required to produce the 424 113 t of olive cake required for the plan based on an average yield from modern, intensive plantations of 9 t ha^{-1} (Beaufoy, 2000) and the ratio of oil mill products defined by La Cal (2010).

Mass allocation in the case of olive cake pellets resulted in over 55% of the environmental impact of olive production being allocated to olive cake production, whereas the production of olive oil is the principal driver for the crop. On the other hand, economic allocation requires a stable price relationship between products (Werner & Richter, 2000). However, this condition is not met in the case of palm kernel shells and olive cake pellets as the price of these products continues to rise. Consequently, allocation by energy content was used as the default method of allocation in this study.

Normalization and summation

For each plan alternative, the emissions and environmental impacts from all plan alternatives were normalized against Irish and regional (EU-27) data in the first instance (Table 5). All environmental pressures were normalized at the same scales (i.e. Ireland or EU, e.g. emissions, land areas, etc.), irrespective of where the pressures arose. In this way, normalized results at the Ireland or EU scale indicate the relative contribution the bioenergy plan(s) make to environmental pressures currently occurring at those scales.

Normalization totals for the impact categories used in the air, climate and water receptors were calculated by multiplying national and regional emissions with characterization factors given in Table 4 to generate normalization totals for each impact category. Normalization data are presented in Table 5.

After normalization, the normalized data were averaged for each of the SEA environmental receptors to provide an environmental impact for each environmental receptor. Prior to summation, the environmental impacts of the bioenergy plan were classified as either positive or negative. Increases in employment and positive effects on landscape (Fig. 2) were considered to be positive environmental effects, whereas emissions to air, water and land and the use of material assets were considered as negative environmental effects. Sequestration of carbon in soil was considered to be a positive effect, whereas release of carbon was considered to be negative as GHGs are released to the atmosphere. Effects of development plans on biodiversity may either be positive or negative. In this bioenergy plan, species richness declined in all plan alternatives and, consequently, both transformation and occupation effects were considered to be negative. Environmental impacts were then summed to provide a net (overall) environmental impact for each part of the environmental chain (field, processing, transport, power station).

Normalization was also carried out according to the geographical area influenced by emissions/impacts from the bioenergy plan. Accordingly, emissions of climate change gases were normalized against global data, emissions of air pollutants were normalized against regional data and all other emissions and impacts were normalized against national data. Global and regional data were subsequently adjusted to an Irish scale using relative differences in population to enable all data to be presented on one scale.

Results

Data on the impacts of the plan alternatives on the environmental receptors prior to allocation are presented in Table 6.

Allocation

Choice of allocation method had a substantial effect on the magnitude of the environmental impacts of both the palm kernel shell and olive cake pellet plan alternatives. However, the environmental impact of co-firing with imported fuels was always greater than that of using native fuels irrespective of allocation method (Fig. 3a and b). The environmental impact of using palm kernel shells was always lower than that of peat irrespective of allocation method and normalization scale. Irrespective of allocation method, the environmental impact of olive cake pellets was greater than that of peat when the

Table 5 Normalization data used in the study together with the sources for the data

	Ireland	EU-27	Global
Climate (t CO_2 eq)	62 317 950(a)	4 089 000 000(b)	4.18 e + 10(c)
Air acidification (t SO_2 eq)	274 400(d)	18 317 399(d)	
Air ecotoxicity (t 1,4 DCB eq)	1 100 000(d)	124 000 000(d)	
Air O_3 formation (t NMVOC eq)	214 691(d)	29 757 420(d)	
Water eutrophication (t P eq)	38 923(e)	1 257 354(e)	
Water freshwater ecotoxicity (t 1,4 DCB eq)	5.21 e 04(e)	5.86 e 06(e)	
Water quantity (million litres)	650 000(f)	214 735 000(f)	
Soil (t waste)	3 397 683(g)	260 777 000(f)	
Biodiversity transformation (species richness)	2.3589 e + 12(h)	9.37 e + 13(h)	
Biodiversity occupation (species richness)	9 494 518 800(h)	376 759 756 800(h)	
Material assets (million km tonnes)	8750(i)	855 636(i)	
Landscape (hectares)	4 200 100(j)	184 852 200(j)	
Employment (people employed)	1 859 599(k)	323 000 000(f)	
Cultural heritage (hectares)	950 000(l)	51 488 200(m)	

(a) McGettigan *et al.* (2010); (b) EEA (2010a); (c) Sleeswijk *et al.* (2008); (d) EEA (2010b); (e) Styles *et al.* (2009); (f) Eurostat (2010); (g) Le Bulloch *et al.* (2009); (h) Schmidt (2008); (i) International Transport Forum (2009); (j) FAOSTAT (2009); (k) http://www.cso.ie; (l) Ward *et al.* (2007); (m) Joosten & Clarke (2002).

Table 6 Impacts of the plan alternatives on SEA receptors prior to allocation

	Palm kernel	Olive cake	*Miscanthus*	Willow	Peat
Field					
Climate	84 101.9	1 211 202.3	60 204.1	50 427	53 697
Air acidification	9431.5	3973.1	288.0	252.0	31
Air ecotoxicity	4.4	4.0	0.8	0.6	216
Air O_3 formation	0.5	0.4	0.1	0.1	66
Water eutrophication	5285	4558.4	233.9	146.9	2
Water ecotoxicity	2126.2	1805.7	76.5	121.5	
Water use		271 886			
Biodiversity transformation	8.0E+9	1.1E+10	2.2E+7	2.2E+7	3.3E+9
Biodiversity occupation	8.2E+8	2.9E+08	6.9E+7	6.9E+7	1.3E+7
Cultural heritage	1332.1				14 097
Landscape	7310.6		8550	8550	9867.9
Employment	18 374	11 856	50.6	33.3	22.3
Processing					
Climate	1 205 374.0	45 907.5			
Air acidification	2682.8	276.7			
Air ecotoxicity	3021.3	147.7			
Air O_3 formation	2881.4	84.1			
Water eutrophication	89.9	2.8			
Water use	7.5E+9				
Employment	18 374	3443			
Transport					
Climate	99 305.9	10 563.6	1382.8	1382.8	457.2
Air acidification	2924.3	175.3	6.7	6.7	1043.5
Air ecotoxicity	2790.0	205.9	16.1	16.1	91.2
Air O_3 formation	2612.7	198.6	16.5	16.5	8.4
Water eutrophication	91.2	0.0	0.6	0.6	0.3
Material assets	115.5	106.0	12.6	12.6	1.5
Employment	129.4	98.9	23	23	8.2
Power station					
Climate					817 630
Air acidification	106.6	590.6	146.9	266.0	965.4
Air ecotoxicity	178.9	767.1	200.0	256.5	828.3
Air O_3 formation	179.2	757.1	198.0	247.4	789.9
Water eutrophication	6.3	26.7	6.9	8.7	27.8
Soil	10 101	41 604	6465	10 910	35 720
Employment	6	6	6	6	

Climate (t CO_2 eq); air acidification (t SO_2 eq); air ecotoxicity (t 1,4 dichlorobenzene eq); air O_3 formation (t NMVOC eq); water eutrophication (t P eq); water ecotoxicity (t 1,4 dichlorobenzene eq); water use (million litres); biodiversity-number-vascular plants; soil (tonnes to landfill); cultural heritage (hectares); landscape (hectares); employment (number employed); material assets (million km tonnes).

effects of the bioenergy plan were normalized against Irish data (Fig. 3a). When normalized against regional data, the environmental impact of using olive cake pellets was greater than that of peat when energy based and mass allocation methods were used but lower than that of peat when economic allocation was used.

Allocation by energy content was used as the default allocation method in this study. All the results described below for olive cake pellets and palm kernel shells use this method of allocation.

Contribution of the different phases of the plan

The net environmental impact of the different phases of the bioenergy plan was calculated by summing the positive and negative environmental consequences associated with each phase (Fig. 3a and b). Irrespective of the normalization scale, biomass production made the largest contribution to the adverse environmental consequences of each of the biomass importation plan alternatives as well as the peat alternative. Processing of

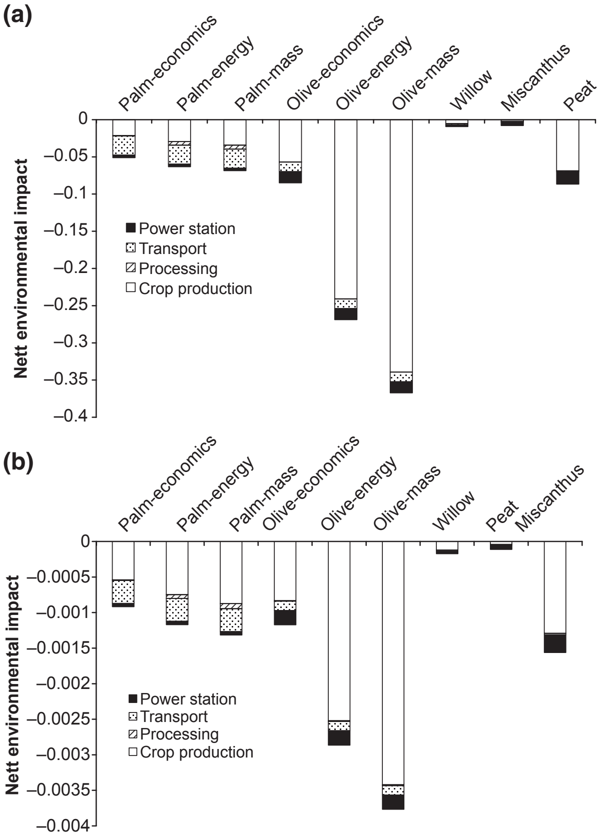

Fig. 3 Summed impacts of the positive and negative aspects of the plan alternatives at different parts of the environmental chain. Normalization was carried out against Irish data (a) and against regional (EU-27) data (b).

biomass prior to transport contributed to the adverse environmental impact of the bioenergy plan in the case of the palm kernel alternative. In contrast, processing operations only had a minor impact in the case of the olive cake alternative and were nonexistent for the willow, *Miscanthus* and peat alternatives where feedstock was taken directly to the power station. The adverse effect of biomass transport operations was greatest in the case of the palm kernel alternative followed by the olive cake alternative and then the willow and *Miscanthus* alternatives. The net environmental impact of factory operations reflected the chemical composition of the feedstock which resulted in emissions to air and deposition of ash to landfill. The environmental impact of factory operations was small compared to that of the other phases of the bioenergy plan in the case of the biomass importation alternatives and the peat alternative.

For all biomass plan alternatives, the negative environmental consequences of the land transformation part of the plan on the climate, water and biodiversity receptors were small compared to the negative consequences of the mature phase of the plan.

Impact of the four plan alternatives on environmental receptors

The impact of each of the plan alternatives on each of the SEA environmental receptors was normalized against Irish national data (Fig. 4a) and regional data (Fig. 4b).

For each of the plan alternatives, the positive environmental effects were very small and almost insignificant compared to the negative environmental effects (Fig. 4a and b). The olive cake alternative generated greatest employment (6021) after allocation followed by the palm kernel shell alternative (4229) due principally to the large number of people involved in harvesting and processing. In contrast, much smaller numbers of people were required for the Irish plan alternatives, 80, 63 and 30 for the willow, *Miscanthus* and peat alternatives, respectively. Landscape effects were proportional to the land area required for each alternative and were all positive as the land areas in question were <27% of the area affected by each plan.

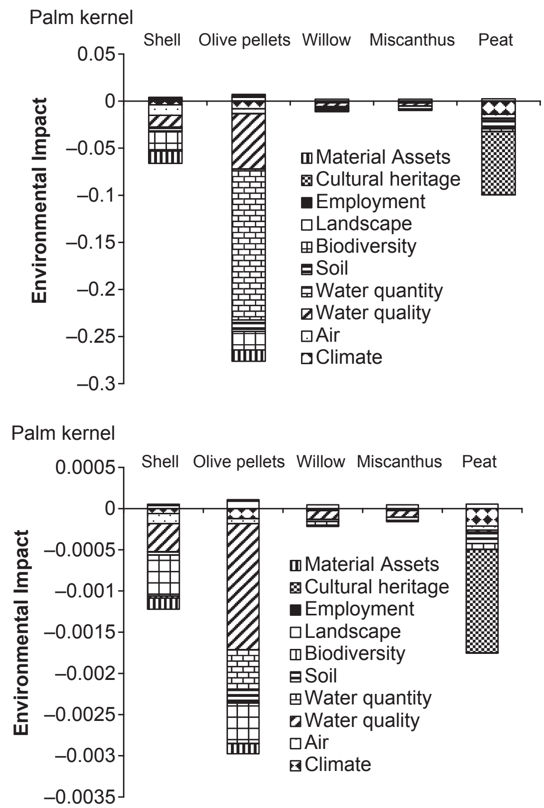

Fig. 4 Impacts of the plan alternatives on each environmental receptor category are shown in this figure. A plan alternative may either have a positive or negative impact on an environmental receptor. Normalization was carried out against Irish data (a) and against regional (EU-27) data (b).

The magnitude of the negative environmental effects was greatest for the olive cake alternative followed by the palm kernel shell alternative. In comparison, the negative environmental consequences of the willow and *Miscanthus* alternatives were much smaller than the plan alternatives which imported biomass feedstock from abroad. The environmental impact of the willow and *Miscanthus* alternatives was also considerably lower than the default alternative of continuing to use peat. The negative environmental consequences of peat usage were greater than those associated with the use of palm kernel shells as a co-firing feedstock while the reverse was true for olive cake pellets. The environmental receptors which were most adversely affected by the bioenergy plan differed between alternatives. Water (quantity and quality) was most adversely affected by the olive cake pellet alternative while biodiversity was the environmental receptor most adversely affected by the palm kernel shell alternative. Water quality was adversely affected by the native biomass plan alternatives. In contrast, the environmental receptors most affected by continued usage of peat were climate and cultural heritage.

Water

Usage of water during irrigation (olive cake) and processing (palm kernel) increased the negative impact of the biomass importation plan alternatives. Water usage substantially increased the negative environmental impact of the olive cake alternative. In contrast most of the impact on water in the palm kernel shell alternative arose from loadings of nitrate, phosphate and herbicide from agricultural operations. The impact of the 'Irish Biomass' alternatives on water were small in comparison to the 'Foreign Biomass' alternatives and consisted solely of loadings to water of nitrate, phosphate and herbicide. The plan alternative with the smallest impact on water was peat due to comparatively minor emissions of N and P to water bodies.

Air

The adverse impacts of the two foreign plan alternative on air were greater than those of the two Irish alternatives due to greater emissions from processing and transport and, in the case of olive cake, greater power station emissions as a result of higher concentrations of N and S in the fuel. Emissions to air were highest from the palm kernel Shell alternative, the largest contribution came from emissions associated with the transportation of biomass from South-East Asia. In contrast, the largest contribution to the air emissions from the olive cake pellet plan alternative came from processing and pelleting.

Climate

Greenhouse gas emissions for the palm kernel and olive cake plan alternatives were lower than the quantities of GHG emitted during the extraction, harvesting, transportation and combustion of peat, no net GHG emissions were assumed to have arisen from the combustion of biomass feedstocks. Most of the emissions from the palm kernel alternative (59%) were associated with the release of methane during the storage of waste products from palm oil processing whereas 36% of GHG emissions arose from biomass transportation. GHG emissions from the olive cake alternative were almost entirely the result of field operations (92%), and primarily attributable to the effects of land use change with only 2% attributable to transport operations. GHG emissions from the use of willow and *Miscanthus* as feedstocks were smaller than all other plan alternatives considered.

Other environmental receptors

The effects of the bioenergy plan on soil were due to the dumping of ash from the power station in landfill. The olive cake alternative had the greatest adverse effect on the soil environmental receptor followed by that of the palm kernel alternative. In comparison, the adverse effects on soil of the Irish biomass alternatives were small. The impact of all plan alternatives on biodiversity arose as a result of the transformation and occupation of land by the new crops. On an annual basis, occupational effects had a greater effect than land transformation effects. The palm kernel and olive cake alternatives had the greatest impacts on biodiversity followed by peat. In comparison the willow and *Miscanthus* alternatives had comparatively smaller effects on biodiversity.

The impact of each plan alternative on each of the SEA environmental receptors was also normalized against EU-27 data (Fig. 4b). Positive environmental impacts were very small in comparison to negative environmental impacts. EU-27 normalization did not change the overall ranking of the plan alternatives. The olive cake alternative had the greatest adverse environmental impact followed by the peat and palm kernel alternatives, respectively. In comparison, the adverse environmental impacts of the two Irish scenarios were small. The negative environmental impact of the biomass importation alternatives was considerably greater than those of the alternatives in which biomass feedstocks were produced locally. The adverse environmental impact of the biomass importation alternatives was dominated by the impact of the plan on water and biodiversity while the adverse environmental impact of the

Irish alternatives was dominated by the effects of the bioenergy plan on water quality. The impact of the peat alternative was greatest on the cultural heritage environmental receptor reflecting the use of a diminishing natural asset (peat).

Effect of normalization scale

Different normalization scales were used to provide a more robust analysis and avoid dependency on one normalization scale particularly where data for certain environmental receptors may be poorly developed or nonrepresentative. For the biomass plan alternatives, changing from an Irish to a regional (EU-27) normalization scale made little change to the relative impacts of the alternatives on each of the environmental receptors (Fig. 4a and b). Similarly, changing normalization scales had little impact on the apportionment of environmental impact according to the stage of the bioenergy plan (Fig. 3a and b). Changing the normalization scale did not change the overall result that the environmental impact of continued usage of peat as being lower than those of olive cake alternative but higher than those of the palm kernel, willow and *Miscanthus* alternatives. When environmental emissions and impacts from each plan alternative were normalized according to their geographical impact (Fig. 5), the analysis confirmed the overall ranking of the alternatives. Overall, the use of different normalization scales showed that the olive cake alternative had the greatest (negative) environmental impact followed by the peat alternative. While the olive cake importation alternative had a greater adverse environmental impact than the default alternative of peat utilization the use of peat had a greater environmental impact than the use of palm kernel shell and either willow or *Miscanthus*.

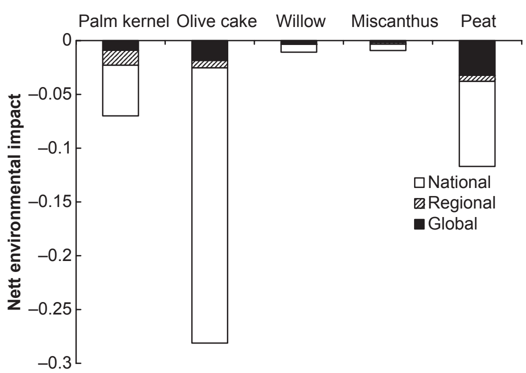

Fig. 5 Summed impacts of the plan when appropriate data was normalized against global (climate), regional (EU-27) (air) and local scales (all other receptors) before being summed according to scale.

Discussion

The use of biomass to generate energy will increase across Europe as EU member states strive to achieve their renewable energy targets by 2020. Member states were required to establish renewable energy targets as a result of the Renewable Energy Directive (2009/29/EC) and bioenergy will play a significant role in renewable energy generation. Biofuels can only count towards renewable energy targets if they can be shown to meet sustainability criteria set out in the directive. Concurrently, member states are also required to conduct a SEA of all plans and programmes likely to have an effect on the environment (2001/42/EC). This study has shown that SEA can be used to structure a comprehensive quantitative assessment which incorporates LCA methodologies to comprehensively compare the environmental and social performance of alternative bioenergy options. This procedure could be of global interest to countries within and outside Europe with ambitious plans to increase renewable energy generation from biomass resources. Application of this procedure to potential bioenergy plans could assist policy makers to select more sustainable bioenergy options for development. We incorporated quantitative LCA methodologies into a SEA framework to quantify the environmental impact of bioenergy plans. From the results of this study, it should be equally possible to incorporate SEA receptors into LCA thus allowing LCA practitioners to widen the environmental scope of LCA.

Normalization is a commonly used tool in environmental analysis in which emissions and impacts for the system under consideration are expressed as a proportion of total environmental loading at national, regional or global scales (Huijbregts & Seppala, 2001; Brentrup *et al*., 2004; Styles *et al*., 2009). Normalization transforms widely different magnitudes of emissions and impacts into comparable values among impact categories (Huijbregts & Seppala, 2001; OECD, 2002; Brentrup *et al*., 2004). Therefore, application of the normalization technique to different impact categories considered within environmental receptors used in this study is a logical step in the comparable quantification of receptors.

The weighting effect that arose from maintaining the SEA receptor structure may be more controversial. Weighting factors may be manipulated to generate desirable results (Kuosmanen & Kortelainen, 2005), and ISO 14040 recommends that the results of weighting are used only for internal analysis and not for public communication. Summation of normalized scores is an important and simple step which is commonly used in environmental quantification and comparison (OECD, 2002; Rudenauer *et al*., 2005; Styles *et al*., 2009). In this study, final results are presented in relation to environmental receptors, and in order to consider all impact categories relevant to each receptor without conveying varying weights to the different receptors, LCA impact categories were given equal weight within receptors and implicit variable weighting across receptors where multiple impact categories were relevant. This approach gives primacy to the SEA structure, and gives equal weight to all environmental receptors defined therein. It could be argued that the different scope, severity and reversibility of different environmental problems should be reflected in different weighting, although this depends on perspective and also possibly geographic location. For example, the Eco-indicator 99 approach defined three archetypes to categorize perspective – hierarchist, egalitarian and individualist (Pre, 1999). The 'passive' weighting applied here is considered to be a practical starting point in the development of the described assessment procedure – the procedure may be elaborated through application of more active and sophisticated weighting, oriented on either impact categories or final receptors, at users' discretion. For examples, weighting factors could be used to prioritize particular objective(s) for a plan, in relation to certain environmental receptors.

In this study, environmental impact of biomass co-products was allocated according to energy content. This approach was adopted as the environmental impact of co-products was exaggerated when mass allocation was used, whereas economic allocation can be unstable owing to sometimes wide variations in the relative prices of the products. However, either mass or economic allocation may also be used and may be more relevant depending on the systems being considered, although in practise no allocation method is ideal. The choice of allocation method had a large effect on the magnitude of the environmental impact of the palm kernel shell and the olive cake pellet scenarios. Similarly, Luo *et al.* (2009) found that allocation method had a large effect on their LCA of corn-stover ethanol. Crucially for our study, however, the choice of allocation method did not change the overall ranking of the plan alternatives, increasing confidence in the robustness of our results.

Pressure on indigenous biomass supplies will lead many countries to consider importing biomass (Woods *et al.*, 2006). However, as bioenergy is driven to a large extent by environmental policy, it is necessary to ensure that bioenergy plans are beneficial, and optimized, from an environmental perspective. LCA of alternative energy carriers and delivery pathways is often used to inform policy on climate change mitigation. Such analyses, however, are often confined to greenhouse gases (GHG–LCA) (Brander *et al.*, 2009). More comprehensive LCA consider eutrophication, acidification, toxicity and even biodiversity, although the analysis becomes more complicated, and the range of commonly use environmental impact categories remains limited. In contrast, SEA considers a wider range of environmental receptors, but often in a qualitative manner. In this study, we employ an approach not dissimilar to that followed in consequential LCA (Brander *et al.*, 2009) to quantify impacts as far as possible across the broad range of environmental receptors considered in SEA, and thus offer a much broader quantitative assessment of the impact of a bioenergy plan on the environment.

This study was based on an Irish government plan to mitigate GHG emissions and peatland destruction by replacing 30% of the power station peat requirement with biomass (Department of Communication, Marine & Natural Resources, 2007). The results showed that the replacement of peat with indigenous biomass (willow and *Miscanthus*) grown in the vicinity of the power stations has the effect of reducing the burden on the environment. In contrast, biomass importation had a considerably more adverse effect on the environment at a global level compared to the use of indigenous energy crops. Transport only contributed in a relatively minor way to the adverse environmental consequences of biomass importation, however. Similar results were reported by Thornley (2008). The effects on the environment likely to arise when biomass is imported occur largely as a result of biomass production. Consequently, while the importing country benefits from the importation of biomass in terms of direct environmental accounting, the exporting country bears the brunt of the environmental damage. Additionally, while GHG emissions are reduced in the importing country, that country is still dependent on an energy carrier which has to be imported from abroad often via long delivery pathway, negating any security of supply benefits associated with indigenous bioenergy.

This study has shown that a quantitative analysis of bioenergy plans can be carried out using a broad range of environmental receptors such as those defined by SEA. Such an analysis can be used to inform policy before critical and far-reaching decisions are taken.

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