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Indicators to support environmental sustainability of bioenergy systems

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ABSTRACT

Indicators are needed to assess environmental sustainability of bioenergy systems. Effective indicators will help in the quantification of benefits and costs of bioenergy options and resource uses. We identify 19 measurable indicators for soil quality, water quality and quantity, greenhouse gases, biodiversity, air quality, and productivity, building on existing knowledge and on national and international programs that are seeking ways to assess sustainable bioenergy. Together, this suite of indicators is hypothesized to reflect major environmental effects of diverse feedstocks, management practices, and post-production processes. The importance of each indicator is identified. Future research relating to this indicator suite is discussed, including field testing, target establishment, and application to particular bioenergy systems. Coupled with such efforts, we envision that this indicator suite can serve as a basis for the practical evaluation of environmental sustainability in a variety of bioenergy systems.

1. Introduction

Indicators to assess the condition of the environment and monitor trends over time are needed to characterize conditions under which resource uses are sustainable. We define environmental indicators as environmental measures (Heink and Kowarik, 2010) that provide information about potential or realized effects of human activities on environmental phenomena of concern. We define environmental sustainability as the capacity of an activity to continue while maintaining options for future generations and considering the environmental systems that support the activity (Bruntland, 1987). Whereas much work has focused on the development of environmental indicators in general, only recently have stakeholders focused attention on developing indicators for sustainable bioenergy systems, and no consensus has yet emerged regarding which indicators should be given the highest priority (Buchholz et al., 2009).

The bioenergy supply chain includes the production or procurement of biomass feedstock, post-production processing and conversion (referred to in this paper as "processing"), and various transportation stages. Beneficial co-products (e.g., distillers grains) and waste by-products (e.g., biorefinery effluent) may be created in different stages of the supply chain. Feedstocks include annual and perennial plants, residues from agriculture, forestry, and related industries, and other organic wastes. The choice of feedstocks is a strong determinant in characterizing a given bioenergy pathway with implications for the applicable set of sustainability indicators.

Bioenergy systems are expected to expand in coming decades for several reasons. First, leaders in many countries view domestic bioenergy systems as more secure and sustainable than imported fossil fuels. Second, economic growth is expected to increase energy demand overall. Third, bioenergy systems are perceived to support rural development and employment. Fourth, technological advances continue to increase the affordability and sustainability of bioenergy. Furthermore, government policies in the United States (U.S.) and Europe call for an expansion of liquid fuel generation and combustion from cellulosic bioenergy feedstock sources, although those feedstocks are not currently in heavy use. The Energy Independence and Security Act of 2007 (EISA) mandates that at least 16 billion gallons (~60.6 billion liters) of cellulosic biofuel be produced annually in the U.S. by 2022 (EISA, 2007). Member states of the European Union aim for biofuel to comprise 10% of their transportation fuel use by 2020, with incentives to encourage cellulosic and other second-generation biofuels (European Parliament and Council, 2009).

As societies increase use of bioenergy, stakeholders are questioning the environmental benefits of bioenergy compared to other energy options. Currently there is disagreement regarding whether bioenergy systems contribute to or ameliorate environmental problems such as depletion of nutrients in soil, erosion,

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runoff of nutrients and toxins, consumptive water use, greenhouse gas buildup, biodiversity loss, air pollution, and productivity loss (Jordan et al., 2007; Keeney, 2008; Williams et al., 2009). Differences of opinion often relate to past land use, crop choice, management practices, processing, and prevailing environmental conditions where the feedstock is grown (Jordan et al., 2007; Robertson et al., 2008; Scharlemann and Laurance, 2008; Kline et al., 2009). In the U.S., much of the debate has focused on the historic effects of conventional crop systems in the Midwest, the source of corn (*Zea mays*) for the majority of current U.S. ethanol production. However, cellulosic bioenergy is often perceived as holding greater opportunity for future sustainability than corn-based ethanol (Robertson et al., 2008; Kline et al., 2009). Because this debate coincides with an expected increase in bioenergy use and because of regulations that require bioenergy to be produced in an environmentally responsible manner, there is a need to characterize conditions under which bioenergy systems can be implemented sustainably (Hecht et al., 2009). This paper presents a set of indicators that can be used to characterize the environmental side of this equation.

The set of environmental indicators selected for assessing the sustainability of different types of bioenergy systems should apply to both large regions and local sites and should be useful to diverse stakeholders. For example, policymakers may focus on sustainability of the entire supply chain, agronomists may recommend sustainable bioenergy feedstock crops and management practices for different locations, and operation managers may seek to improve their feedstock production and processing systems. Indicators may also help in the implementation of certification programs (several are already in development) that can be applied throughout the supply chain or to its components (van Dam et al., 2008).

Although much work is still needed to identify, test, and implement a small set of environmental indicators that is useful to the diverse stakeholders involved in bioenergy systems, progress has been made. Sustainability attributes of agricultural practices in general have been discussed and defined by the Millennium Ecosystem Assessment (MEA, 2005), the National Sustainable Agriculture Information Service (Sullivan, 2003; Earles and Williams, 2005), and Dale and Polasky (2007). In addition, several national and international efforts are underway to select sustainability indicators for bioenergy, including the Roundtable on Sustainable Biofuels (RSB, 2010), U.S. Biomass Research and Development Board, Global Bioenergy Partnership (GBEP, 2010), and Council on Sustainable Biomass Production (CSBP, 2010). The preliminary suites of indicators arising from these efforts are diverse, and the differences among them are important, but here we note two broad characteristics. First, these suites tend to include numerous, broadly defined indicators. Second, many of the indicators in these suites tend to focus on assessments of management practices and their predicted environmental effects rather than on measurements that relate to realized environmental effects. These approaches have advantages. Large numbers of broad indicators can in principle capture a wide range of environmental effects. Also, assessing management practices may often be less expensive than making empirical measurements; indeed, simple measurements of some effects, such as tropospheric ozone formation, may not be feasible with respect to particular bioenergy systems. On the other hand, measuring large numbers of indicators can be prohibitively expensive (NRC, 2008a). Furthermore, current understanding of the effects of bioenergy management practices on the environment is limited, especially for systems not yet in wide use, such as cellulosic bioenergy. Therefore a need remains for a small set of concrete indicators that focus on realized environmental effects of bioenergy systems.

This paper identifies a suite of 19 indicators selected to collectively characterize important effects that many bioenergy systems have or are likely to have on environmental sustainability. The suite is organized according to six categories: soil quality, water quality and quantity, greenhouse gases, biodiversity, air quality, and productivity. These categories were selected to reflect the major areas of environmental concern surrounding bioenergy systems. They are also similar to categories used by national and international efforts working to establish suites of sustainability indicators for bioenergy. For each category, we discuss the relationship of proposed indicators to ecosystem properties and address measurement considerations. After presenting indicators in each category, we discuss future research directions, applications of these indicators to specific bioenergy systems, and interpretation of these indicators. This paper provides a basis for other researchers and investigators to move forward to evaluate and implement environmental indicators for bioenergy systems.

Table 1
Criteria for selection of useful environmental indicators.

Are easily measured
Are sensitive to stresses on system
Respond to stress in a predictable manner
Are anticipatory: signify an impending change in the environmental system
Predict changes that can be averted by management actions
Are integrative: the full suite of indicators provides a measure of coverage of the key gradients across the environmental systems (e.g., soils, vegetation types, and temperature)
Have a known response to natural disturbances, anthropogenic stresses, and changes over time
Have known variability/spread in response to given environmental changes

Adapted from Dale and Beyeler (2001).

2. Approach

Where feasible, indicators were selected to empirically measure environmental effects rather than to infer such effects through assessment of management practices. In some cases, however, models based on management practices are the only feasible way to estimate the environmental effects of bioenergy systems (e.g., greenhouse gas fluxes or secondary particulate formation, discussed in Sections 3.3 and 3.5, respectively).

Our selection of indicators was based on research in the disciplines related to each category of indicators, on other efforts to select sets of indicators, and on previous work describing criteria for selecting useful indicators [e.g., Dale and Beyeler (2001), Table 1]. The diversity of indicators needed to broadly assess environmental sustainability may not allow for a uniform, well-defined indicator selection process (NRC, 2008a); therefore, expert judgment is an important part of the selection process. Collectively, the proposed suite of indicators forms a hypothesis of how environmental effects of bioenergy systems may be assessed, and that hypothesis needs to be tested in diverse bioenergy systems.

3. Categories of indicators

3.1. Indicators of soil quality

Among the environmental systems for which indicators have been chosen, soils are especially important because soil quality affects the broader ecosystem, the immediate productivity of bioenergy crops, and the maintenance of productive capacity for future generations. Our selection of soil indicators was influenced by prior research on soil indicators in general (Doran and Parkin, 1996; Garten et al., 2003; Karlen et al., 2003; Pattison et al., 2008; Adair et al., 2009) as well as on agronomy research focused on bioenergy crops in particular (Mann and Tolbert, 2000; Tolbert et al., 2002; Moscatelli et al., 2005; Garten et al., 2010).

Table 2
List of recommended environmental indicators for bioenergy sustainability, along with associated management pressures and environmental effects expected to be captured by each indicator.

Category	Indicator	Units	Related management pressures	Potential related environmental effects	Reference that discusses methods used to collect data
Soil quality	1. Total organic carbon (TOC)	Mg/ha	Crop choice, tillage	Climate change, N mineralization, humification, water holding capacity, infiltration, CEC	Doran and Jones (1996)
	2. Total nitrogen (N)	Mg/ha	Crop choice, tillage, N fertilizer application, harvesting practices	Eutrophication potential, N availability	Bremner and Mulvaney (1982)
	3. Extractable phosphorus (P)	Mg/ha	Crop choice, tillage, P fertilizer application, harvesting practices	Eutrophication potential, P availability	Olsen et al. (1954) and Mehlich (1984)
	4. Bulk density	g/cm^3	Harvesting practices, tillage, crop choice	Water holding capacity, infiltration, crop nutrient availability	Doran and Jones (1996)
Water quality and quantity	5. Nitrate concentration in streams (and export)	Concentration: mg/L; export: kg/ha/year	Crop choice, % of residue harvested, tillage, N fertilizer application	Eutrophication, hypoxia, potability	Eaton et al. (2005)
	6. Total phosphorus (P) concentration in streams (and export)	Concentration: mg/L; export: kg/ha/year	Crop choice, % of residue harvested, tillage, P fertilizer application	Eutrophication, hypoxia	Eaton et al. (2005)
	7. Suspended sediment concentration in streams (and export)	Concentration: mg/L; export: kg/ha/year	Crop choice, % of residue harvested, tillage	Benthic habitat degradation through siltation, clogging of gills and filters	Eaton et al. (2005)
	8. Herbicide concentration in streams (and export)	Concentration: mg/L; export: kg/ha/year	Crop choice, herbicide application, tillage	Habitat degradation through toxicity, potability	Eaton et al. (2005)
	9. Peak storm flow	L/s	Crop choice, % of residue harvested, tillage	Erosion, sediment loading, infiltration	Buchanan and Somers (1969)
	10. Minimum base flow	L/s	Crop choice, % reside harvested, tillage	Habitat degradation, lack of dissolved oxygen	Buchanan and Somers (1969)
	11. Consumptive water use (incorporates base flow)	Feedstock production: m^3/ha/day; biorefinery: m^3/day	Crop choice, irrigation practices, downstream biomass processing	Availability of water for other uses	Feedstock production: calculated from flow measurements. Biorefineries: reported total water withdrawn used as proxy
Greenhouse gases	12. CO_2 equivalent emissions (CO_2 and N_2O)	kg C_{eq}/GJ	N fertilizer production and use, crop choice, tillage, liming, fossil fuel use throughout supply chains	Climate change, plant growth	Spreadsheet models (e.g., GREET; Wang, 2002), with various submodels
Biodiversity	13. Presence of taxa of special concern	Presence	Crop choice, regional land uses, management practices	Biodiversity	Various methods exist depending on taxa selected
	14. Habitat area of taxa of special concern	ha	Crop choice, regional land uses	Biodiversity	Various methods exist depending on taxa selected; for one approach see: Turlure et al. (2010)
Air quality	15. Tropospheric ozone	ppb	Fossil fuel use in production and processing, quality and mode of combustion of biofuel	Human health, plant health	Combination of sources and methods necessary, for example: EPA Mobile Source Observation Database, Community Multiscale Air Quality model (for example: Appel et al., 2007), reports from biorefineries, collation of vehicle use with emissions data per fuel type (for example: Gaffney and Marley, 2009)
	16. Carbon monoxide	ppm	Fossil fuel use in production and processing, mode of biofuel combustion	Human health	
	17. Total particulate matter less than 2.5 μm diameter ($PM_{2.5}$)	$\mu g/m^3$	N fertilizer application, fossil fuel use in production and processing, mode of biofuel combustion	Visibility, human health	
	18. Total particulate matter less than 10 μm diameter (PM_{10})	$\mu g/m^3$	Fossil fuel use in production and processing, other agricultural activities, solid biomass combustion	Visibility, human health	
Productivity	19. Aboveground net primary productivity (ANPP)/yield	g C/m^2/year	Crop choice, management practices	Climate change, soil fertility, cycling of carbon and other nutrients	Grasslands: Scurlock et al. (2002). Forests: Clark et al. (2001)

Four indicators of soil quality are recommended: total organic carbon, total nitrogen, extractable phosphorus, and bulk density (Table 2). These indicators were selected based on their ability to reveal changes in soil properties as a function of bioenergy crop management, including carbon balance, nutrient availability and mineralization, cation exchange capacity (CEC), humification, microbial community dynamics, erosion, leaching potential, soil porosity, and soil water holding capacity.

Total organic carbon (TOC) is often seen as the most important indicator of soil quality (Reeves, 1997). TOC integrates a wide range of important soil properties and functions and also is a direct cause of several positive soil responses. First, it serves as the primary source of energy for soil microbial communities, which, in turn, promote crop growth by supporting nitrogen mineralization (NRCS, 2009). Second, high TOC suggests high humus levels, which promote water holding capacity, infiltration, and CEC. Third, compounds in soil organic matter, which correlates with TOC, help bind soil aggregates in non-calcareous soils, contributing to porosity and further enhancing water holding capacity and infiltration (NRCS, 2009).

In addition to the role of TOC as an indicator in assessing soil quality, accurate measurements of soil carbon are also important in estimating carbon dioxide flux associated with bioenergy systems, as discussed in Section 3.3. Soil carbon changes are likely to occur because of land-use changes associated with the initial implementation of bioenergy systems, as well as during the ongoing operation of those systems.

Total nitrogen (N) and extractable phosphorus (P) measure the two most important soil nutrients in typical productive land management systems. Most N in soil is bound in organic compounds and is not available to plants. However, total N is considered a valid indicator because N mineralization is driven by the availability of organic N in the soil, so that plant-available N (ammonium and nitrate) is closely related to total N (Vlassak, 1970). Excessive soil N and P can result in nutrient runoff and leaching, leading to downstream eutrophication. In addition, excess soil nitrate may increase N volatilization as the potent greenhouse gas nitrous oxide (Dalal et al., 2003; Snyder et al., 2009). Conversely, depletion of soil N and P threatens the future productivity of soil.

Finally, bulk density is recommended as a physical indicator of soil quality. Bulk density can rapidly be affected by human agronomic practices (Unger and Kaspar, 1994). Bulk density is especially of concern in forestry, because tree harvesting activities can cause soil compaction (Hatchell et al., 1970). Increases in bulk density are usually considered harmful (Unger and Kaspar, 1994), but in some crops, such as switchgrass (*Panicum virgatum*), it is desirable to have light surface soil compaction before sowing in order to improve seed–soil contact (Monti et al., 2001).

Techniques for measuring TOC and bulk density can be found in Doran and Jones (1996). Techniques for measuring total N can be found in Bremner and Mulvaney (1982). Mehlich (1984) and Olsen et al. (1954) describe techniques for measuring extractable P in acidic and calcareous soils, respectively. The appropriate depth of measurement for soil indicators depends on depth of soil layers and cultivation practices on a given site and should remain constant over time.

3.2. Indicators of water quality and quantity

The properties of water in streams draining bioenergy croplands or forest stands influence the ecosystems within and downstream from those streams. Indicators based on water properties can be used to assess whether the agricultural aspects of bioenergy production allow for the maintenance of soil quality, aquatic ecosystems, and clean and plentiful water for human use. Water indicators are affected by some of the same pressures that influence soil indicators (e.g., fertilizer application and vegetative cover). In contrast to soil indicators, water indicators can change more rapidly and integrate changes over an entire watershed, thereby allowing for finer temporal resolution and broader spatial integration of relevant effects. In this sense water quality and quantity reflect the diversity of environmental conditions and land practices that occur upstream and upslope as well as in the past. For example, runoff attributes are influenced by current and past land cover, chemical applications, and soil conditions.

Seven indicators of water quality and quantity are recommended: stream concentrations of nitrate, total phosphorus, suspended sediment, and herbicides; peak storm flow; minimum base flow; and consumptive water use (Table 2). These indicators were selected based on their ability to reveal changes in several environmental properties that might occur as a result of bioenergy crop management: water availability, water potability, aquatic biodiversity, eutrophication, dissolved oxygen, soil erosion, sediment loading, soil leaching potential, soil porosity, and soil water holding capacity. In selecting these indicators, we assume that in most cases, water from feedstock production sites will drain into streams (some of which may be only ephemeral) before reaching lakes, estuaries, or other lentic waters.

Concentrations of nitrate and total phosphorus (P) in streams are indicators of potential eutrophication. Whereas aquatic systems respond to nitrogen (N) in other forms, nitrate is usually the most abundant form, relatively inexpensive to measure, highly mobile, and expected to be sensitive to the management of bioenergy feedstock systems. Furthermore, nitrate in drinking water is also associated with health risks such as methemoglobinemia (Ward et al., 2005). In streams, total P includes dissolved phosphate, organic phosphorus, and phosphate sorbed to suspended sediment. Measurement of total P in streams is especially important during storm events, because P export during storm events tends to dominate watershed P export and is sensitive to crop management practices (Sharpley et al., 2008).

Recent meta-analyses suggest that lotic, lentic and coastal marine ecosystems are generally responsive to both N and P (Francoeur, 2001; Elser et al., 2007). Environmental effects of eutrophication were reviewed by Smith et al. (1999) and are characterized by increased biomass of algae, periphyton, and/or phytoplankton, decreased dissolved oxygen, and death of fish and other animals. In the U.S., the contributions of N and P export to hypoxia in the Gulf of Mexico are of particular concern (Alexander et al., 2008; Dale et al., 2010a).

Concentration of herbicides in streams measures exposure of aquatic life to these chemicals and their potentially toxic effects. Most pesticide use in the U.S. consists of herbicides. In 2000 and 2001 combined, 62% of conventional pesticides used (by mass of active ingredient) consisted of herbicides (Kellogg et al., 2000; Kiely et al., 2004). Schäfer et al. (2007) found that various pesticides, including herbicides, were detrimental to stream macroinvertebrate community structure and ecosystem function when they occur at concentrations lower than those previously known to have such effects. Measuring herbicide concentrations is expensive, and therefore we recommend that only herbicides known to be used or of concern in a given area should be measured.

Suspended sediment concentration is an indicator of stream habitat quality. Siltation diminishes interstitial space in stream substrata, impairs fish spawning grounds, and reduces the ability of sessile benthic organisms to attach to streambeds. Increased turbidity reduces the ability of benthic plants and attached algae to photosynthesize. Reduced benthic productivity and biodiversity can reduce available food for grazing organisms. Suspended sediment also clogs the gills of fish and hinders nutrient uptake by filter feeders. These and other effects of sediment load in lotic environments were reviewed by Wood and Armitage (1997). In addition to its adverse effects on aquatic habitat, suspended sediment also serves as an indicator of soil erosion, which can be used to assess the sustainability of bioenergy systems (Smeets and Faaij, 2010).

In addition to concentrations of nitrate, total P, herbicides, and sediments, export levels per unit watershed area of these substances are also important. Whereas concentrations are indicators of the effects these substances may have on the streams in which they are measured, export levels are related to the effects of these substances on downstream bodies of water (e.g., hypoxia in the Gulf of Mexico or propagation of sediment downstream during flushing events). Area-specific export levels can be calculated by multiplying stream concentrations of each substance by flow measurements and dividing by total watershed area. Because estimating watershed area is straightforward and flow measurements are recommended as indicators in the following paragraph, we do not treat these area-specific export levels as separate indicators.

Two flow properties, peak storm flow and base flow, are indicators of environmental effects of changes in soil and crop hydrologic processes. Base flow is related both to availability and quality of aquatic habitat and to the availability of water for human use. These two issues are considered separately. Interpreting flow measurements requires also measuring rainfall on similar timescales in order to separate the effects of rainfall from those resulting from changes in soil and crop hydrologic properties.

Increased peak flow during storm events can be caused by decreased infiltration and water holding capacity in soil. High peak flows during storms can increase erosion (de Lima et al., 2003) and sediment loading (Lawler et al., 2006). In addition, high peak flows can reduce benthic organism biomass and habitat as a result of streambed scouring and can contribute to potential flood damage downstream.

As an indicator of water quality, base flow should be considered at its minimum, often occurring in summer or early fall, because lotic habitat quality can be limited by minimum base flow (Bunn and Arthington, 2002). During periods of low base flow, dissolved oxygen levels in streams are usually at their lowest due to lower rates of oxygen diffusion into water from the atmosphere and greater depletion of available oxygen supplies in water from respiration by aquatic organisms. Very low dissolved oxygen levels can lead to stress or death of some aquatic organisms, particularly fish.

In addition to its utility as an indicator of lotic habitat quality, base flow also serves as one of two measures of consumptive water use, the seventh recommended water-related indicator. Consumptive water use in bioenergy systems, mostly during feedstock production and in biorefineries, may affect the amount of water available for other human uses (Berndes, 2002; de Fraiture et al., 2008; Stone et al., 2010). Changes in base flow can reflect consumptive water use in feedstock production. For this purpose, base flow should be considered throughout the growing season. It should also be measured sufficiently downstream to capture both irrigation return flow (Huffaker, 2010) and the surface discharge of groundwater sources drawn upon by deep-rooted crops.

Water withdrawn from public sources is recommended as an indicator reflecting consumptive water use in biorefineries (NRC, 2008b). Most consumptive water use in biorefineries consists of evaporation from cooling towers and dryers/evaporators during distillation (NRC, 2008b; Wu et al., 2009). Total water withdrawal is typically metered and easily reported by biorefinery managers. Not all water withdrawn represents consumptive use; however, the extent to which water withdrawal overestimates consumptive use is decreasing as water recycling in biorefineries increases (NRC, 2008b). Consumptive water use in biorefineries can be locally intense (NRC, 2008b).

Standard methods for measuring nitrate, total P, suspended sediment, and several common herbicides can be found in Eaton et al. (2005). Techniques for measuring stream flow can be found in Buchanan and Somers (1969) and Hudson (1993).

3.3. *Indicator of greenhouse gas flux*

Estimated net carbon equivalent (C_{eq}) flux to the atmosphere is recommended to measure the effect of bioenergy systems on atmospheric concentration of greenhouse gases that contribute to climate change (IPCC, 2007) (Table 2). The direct and indirect environmental effects of elevated atmospheric C_{eq} concentrations differ regionally, but, because the atmosphere is well-mixed, those effects do not depend on the locations of C_{eq} release or sequestration. Therefore, C_{eq} release and sequestration throughout the bioenergy supply chain can be summed, and the marginal environmental effects of those fluxes can be estimated using standard global climate models. Hansen et al. (2006) and McMichael et al. (2006) discuss the expected effects of increasing greenhouse gas concentrations on climate, environment, and human health, such as increases in temperature, sea level, extreme weather events, species loss, and disease.

To estimate net C_{eq} flux associated with bioenergy, we recommend that nitrous oxide (N_2O) flux and carbon dioxide (CO_2) flux be considered. N_2O is emitted directly from soil during both nitrification and denitrification (Bouwman et al., 2010), as well as indirectly when volatilized nitric oxide and nitrogen dioxide (NO_x) and ammonia (NH_3) are deposited offsite and converted to N_2O or when leached nitrate is denitrified in waterways (Adler et al., 2007). In agricultural systems, N_2O emissions are strongly dependent on the amount of N fertilizer applied to the soil (Crutzen et al., 2008). In addition to application-related emissions, N_2O is also released, typically in smaller amounts, during the production of nitrate fertilizers, specifically during the intermediate step of nitric acid production (Snyder et al., 2009).

The bioenergy supply chain also contains several sources and sinks for CO_2 that must be considered in estimating net greenhouse gas flux. Where feedstocks are produced, these sources and sinks include changes in carbon stocks in biomass and soil, dissolution of agricultural lime, and fossil fuel used in sowing, tilling, harvest, and application of soil inputs. Offsite sources upstream from feedstock production include fossil fuel used in the manufacture and transport of agricultural inputs such as fertilizer, pesticide, seed, and agricultural lime. Offsite sources downstream from feedstock production include fossil fuel used in processing (such as at biorefineries) and in the transportation of feedstock and fuel. In addition, electricity must be generated off-site for use in all stages of the supply chain. This list of sources and sinks is an extension of that used by West et al. (2010) for agriculture. The exclusion from this list of carbon fixed in photosynthesis or released through the oxidation of biomass is consistent with the assumption of other researchers (e.g., West et al., 2010) that any difference between these two quantities is represented by changes in soil or standing biomass carbon stocks.

Estimated values for these various sources and sinks of N_2O and CO_2 can be collected and summed using the life cycle assessment (LCA) approach. Standard and useful tools for LCA are multidimensional spreadsheet models such as the GREET (Greenhouse gases, Regulated Emissions, and Energy use in Transportation) and GHGenius software models, which are designed to address full fuel cycle (or well-to-wheels) effects (Wang, 2002; Stanciulescu and Fleming, 2006). These spreadsheet models have advantages in that they are user-friendly, publicly available, straightforward, and relatively transparent. By default, such spreadsheet models often have built-in statistical submodels that can be retained or overridden with measured values or with the results of more sophisticated, external submodels. This flexibility allows users simultaneously to take advantage of information relevant to a given problem and to make use of standard estimates where problem-specific information is not available.

Some default values in spreadsheet models are best replaced with empirical measurements where available. For example, soil carbon measurements are recommended as an environmental indicator of sustainability in part because they relate not only to several aspects of soil quality but also to greenhouse gas flux. Assuming soil carbon measurements are made, the accuracy of site-specific LCAs can be improved by substituting those measurements for statistically modeled estimates in spreadsheet models.

Default emission factors in spreadsheet models for N_2O released from soil can be replaced with empirical measurements or with more sophisticated models when appropriate data are available. Default factors may be based on straightforward statistical models that estimate N_2O emissions from N fertilizer application rate alone (Wang et al., 2008). Such approaches are appropriate for global emissions but fail to capture important site- and management-specific variations in the relationship between applied N and N_2O flux (Del Grosso et al., 2010). Ideally, local N_2O emissions are measured empirically, but the two common methods for measuring N_2O emissions face practical challenges: eddy covariance towers (e.g., Eugster et al., 2007) are expensive to establish and maintain, and chamber measurements are also expensive when enough chambers are used to detect the effects of "hotspots," small areas with high N_2O emissions compared to surrounding soil (Neftel et al., 2007; Hellebrand et al., 2008). Because of these challenges, models are often used to estimate soil N_2O flux from agronomic systems, including bioenergy production (Adler et al., 2007; Bouwman et al., 2010). The simulation model DAYCENT (Parton et al., 1998) has been used to estimate soil N_2O flux from various bioenergy crops, using as inputs daily weather simulations, soil texture and hydraulic properties, crop growth dynamics, N application rate, harvest schedule, and tillage (Adler et al., 2007). However, modeling of N_2O emissions faces "tremendous challenges" because the potentially confounding influences and interactions of several factors (such as the pore space characteristics, bulk density, temperature, pH, and carbon content of soil) are not well understood (Farquharson and Baldock, 2008). As data become more widely available, measurements should be used to validate modeled estimates of N_2O flux (e.g., Del Grosso et al., 2010).

In addition to CO_2 and N_2O, methane (CH_4) can be important in calculating C_{eq} emissions. In bioenergy systems, CH_4 is emitted primarily when solid biomass is burned on small scales, such as for domestic cooking and heating, or when open biomass burning is a part of feedstock production. In these cases CH_4 may be a small but significant contributor to C_{eq} flux, contributing 14% or less of total combustion-related C_{eq} emissions (Yevich and Logan, 2003; Ito and Penner, 2004; Macedo et al., 2008). Changes in land management may alter the balance of methanogenesis and methanotrophy in soil, but such changes typically do not affect the C_{eq} balance of bioenergy systems as much as do changes in CO_2 and N_2O fluxes (Ussiri et al., 2009; Cherubini, 2010; Shurpali et al., 2010).

Estimates of net C_{eq} flux from bioenergy systems based on LCAs differ, even for similar systems. Reviews of greenhouse gas LCAs for bioenergy have sought to identify sources of those differences (Liska and Cassman, 2008; Cherubini et al., 2009; Davis et al., 2009; Gnansounou et al., 2009). Differences in system boundaries were important (e.g., inclusion of co-products and use of economic models to attempt prediction of indirect land-use changes). Most reviews also cited differences in the treatment of reference conditions (i.e., displaced fossil fuel systems). Such methodological challenges compound challenges in accurately estimating components of C_{eq} flux, such as soil carbon and N_2O emission from soils. Despite these difficulties, the openness and flexibility of greenhouse gas LCAs makes them an appropriate tool for different stakeholders to evaluate and compare the C_{eq} flux of different bioenergy systems.

3.4. Indicators of biodiversity

Measures of biodiversity are valuable indicators of sustainability in agroecosystems (Biala et al., 2005). Biodiversity can relate to any type of organism, including plants, animals, fungi, and microbes. Biodiversity indicators are useful in comparing different agricultural systems because, in addition to being valued for its own sake, biodiversity is affected by other environmental changes such as erosion, nutrient loss, and land-use change. Bioenergy systems are likely to affect biodiversity in several ways. For example, feedstock cultivation in extensive monocultures or pollution from biorefineries may cause loss of species, changes in abundance of species, and habitat degradation or loss. By contrast, appropriately managed perennial bioenergy cropping systems can improve habitat for some species, such as grassland birds (Murray et al., 2003). For the purpose of selecting biodiversity indicators, we focus on the direct effects on biodiversity of land-use changes involved in the production or procurement of feedstocks because those effects are likely to be measurable in the short term and can be spatially extensive.

The presence and habitat area of taxa of special concern are recommended as indicators to measure the effects of bioenergy systems on biodiversity (Table 2). The actual taxa that are of special concern vary in identity and number by site and region. Examples include rare native species, biodiversity-related keystone species, and taxa that are part of bioindicators. These three examples are defined and discussed below. Other taxa of special concern include species of commercial value, cultural importance, or recreational value.

Native species that are locally or globally rare (whether naturally or through human activity) or that could become rare due to bioenergy system implementation are examples of taxa of special concern. Rare or potentially rare species may be at greater risk of extinction (local or global) than common species; therefore, monitoring their presence may lead to a relatively larger probability of capturing a decrease in biodiversity due to their extirpation. In an effort that focused on rare species at risk [using the definition of Master (1991)], Lawler et al. (2003) found that habitat of at-risk species correlated well with the habitat of other species in the Middle Atlantic region of the U.S., thus serving as an indicator of biodiversity beyond the at-risk species themselves.

Biodiversity-related keystone species are another example of taxa of special concern. Power et al. (1996) defined a keystone species as "one whose impact on its community or ecosystem is large, and disproportionately large relative to its abundance." Power et al. (1996) explained that "impact" can be defined with respect to various ecosystem traits. Here we are interested in species with disproportionate effects on biodiversity, such as the gopher tortoise (*Gopherus polyphemus*) in the southeastern U.S., whose burrows provide habitat for a large number of other species (McCoy and Mushinsky, 2007), or other ecosystem engineers such as prairie dogs (*Cynomys* spp.) in arid grasslands (Bangert and Slobodchikoff, 2006; Shipley and Reading, 2006). The impact of the loss of such a species from an ecosystem can be amplified by the resultant loss of other species.

Other taxa of special concern are those that comprise what are commonly termed "bioindicators," which are taxa frequently used to monitor the condition of an environment or ecosystem. Bioindicators often consist of aquatic taxa and are used to assess the impacts of anthropogenic stresses on water quality. The presence of some taxa in aquatic systems downstream from bioenergy feedstock production may indicate positive effects of bioenergy systems (e.g., if bioenergy land management results in less chemical or sediment loading than prior land use). The presence of other taxa may indicate negative effects of bioenergy (e.g., if crops require more fertilizers, herbicides or pesticides than prior land use).

In addition to aquatic organisms, other generalizations can be made about types of taxa likely to be affected by bioenergy systems, even though the selection of particular indicator taxa is inherently site- or region-specific. Organisms likely to be affected include aquatic animals, arthropods (Gardiner et al., 2010), birds, small mammals, and ground flora (Semere and Slater, 2007).

For many species of special concern, it is more feasible to measure the extent of suitable habitat than to measure the presence or abundance of a taxon directly. For example, Turlure et al. (2010)

demonstrated the validity of using habitat area as a proxy for population size for two vulnerable peat bog butterflies. By showing that habitat area worked best as a proxy when defined according to functional resources rather than host plants, their study emphasized the importance of carefully defining suitable habitat. Because species of special concern in different systems differ widely in habit, methods for measuring presence and habitat area of those taxa also differ.

3.5. Indicators of air quality

Most air pollutants resulting from bioenergy use derive directly or indirectly from combustion in feedstock production and processing as well as in final use (e.g., powering vehicles by burning liquid biofuels). Carbon monoxide, tropospheric ozone, and two fractions of suspended particulate matter (PM_{10} and $PM_{2.5}$) are recommended as indicators to measure the effects of bioenergy on air quality (Table 2).

Almost all carbon monoxide (CO) emissions related to bioenergy derive from combustion. Combustion throughout the bioenergy supply chain includes combustion of biofuels for vehicles, heat, and electricity, as well as the combustion of fossil fuels used in the production of bioenergy. However, CO emissions from cars and other transportation sources have been virtually eliminated with the advent of the catalytic converter in the 1970s and replacement of the legacy fleet. CO is a minor contributor to climate change, but it is of environmental concern primarily for two reasons. First, it has severe effects on human health in high concentrations and may also be harmful at low, chronic concentrations (Townsend and Maynard, 2002; Chen et al., 2007). Second, it is a precursor to ozone production, as discussed below. The emission of CO in biofuel combustion varies widely based on fuel type and combustion method. In some cases, an increase in the overall efficiency of a combustion process can have a counterintuitive inverse relationship with CO emissions (Venkataraman and Rao, 2001). Because present-day liquid biofuels are oxygen-containing compounds, burning biofuel either as an additive to petroleum products or as a primary fuel can result in lower CO emissions than burning pure gasoline or petroleum diesel fuel.

Tropospheric ozone is an important pollutant and is also associated with smog and haze. Ozone can aggravate or damage the respiratory system and can also damage vegetation, potentially reducing crop yields and biodiversity. Tropospheric ozone is formed by the reaction of nitric oxide and nitrogen dioxide (NO_x) with non-methane organic gases (NMOGs) (Atkinson, 2000) or with CO (NRC, 1977). These compounds are emitted in varying amounts from all combustion processes involved in the production and use of bioenergy. NO_x is particularly associated with distillation processes for ethanol production. The reaction of these ozone precursors may occur far from emission sources; therefore, NO_x associated with bioenergy may react with NMOGs or CO from unrelated sources or vice versa. Ambient air quality standards for ozone in the U.S. (EPA, 2010) have been growing stricter, and many regions, mostly urban, have entered or will enter non-attainment status for ozone. Thus, any effect of bioenergy production or use on ambient ozone levels will be closely monitored by regulators.

$PM_{2.5}$ measures mass per unit volume of all airborne particles less than 2.5 μm in diameter, also known as the fine particle fraction. Fine particles can be emitted directly from point sources; such particles (soot, for example) are called "primary" (Seinfeld and Pankow, 2003). Fine particles such as ammonium nitrate, ammonium sulfate, and secondary organic aerosols (SOA) are formed in the atmosphere from gaseous emissions and are known as "secondary" (Seinfeld and Pankow, 2003). Bioenergy systems can contribute to fine particulate pollution through solid biomass combustion or through the emission of various secondary particulate precursors through biofuel combustion (i.e., NMOGs leading to SOA), through burning of fossil fuels during feedstock production or processing [i.e., oxides of sulfur (SO_x), NO_x], or from soil biochemical processes during feedstock production (i.e., ammonia). Fine particles are associated with increased mortality due to lung cancer, cardiopulmonary disease, and other factors (Pope et al., 2002). This association with increased mortality is especially strong for fine particles associated with combustion (Laden et al., 2000). Because the diameters of fine particles in the atmosphere are close to the wavelengths of visible light, fine particles also scatter light effectively and typically reduce visibility more than larger particles (Malm, 1999).

PM_{10} measures mass per unit volume of all airborne particles less than 10 μm in diameter and thus includes those particles measured by $PM_{2.5}$. In addition to fine particles, PM_{10} includes coarse particles, those between 2.5 μm and 10 μm in diameter. Agricultural systems can affect this coarse fraction through tilling and solid biomass combustion (Aneja et al., 2009). As with the fine fraction, the coarse fraction can affect human respiratory health, though health effects may be restricted to the short term (Brunekreef and Forsberg, 2005). Coarse particles also impair visibility, though also to a lesser extent than fine particles (Malm, 1999). The lesser environmental concerns relating to coarse particles, as well as the confounding inclusion of both fine and coarse particles in PM_{10}, are drawbacks to using PM_{10} as an indicator of environmental aspects of bioenergy sustainability. Nonetheless, we recommend its use for two reasons. First, the coarse fraction may have greater influence on health and visibility issues where it dominates the fine fraction in abundance, such as on feedstock production sites and where solid biomass is burned. Second, because of historical Environmental Protection Agency (EPA) regulations in the U.S., more infrastructure exists to measure PM_{10} than to measure $PM_{2.5}$; therefore, even where the fine fraction is of primary concern, PM_{10} may serve as a rough but affordable proxy measure of the fine fraction.

Methods for measuring CO, tropospheric ozone, $PM_{2.5}$, and PM_{10} vary by location. Extensive ambient air monitoring networks have been installed in many regions of the U.S. (AIRNow, 2010) as well as in Europe. The U.S. EPA requires large emitters such as biorefineries to report emissions of some pollutants. Feedstock producers can report equipment usage, which can be combined with data sources such as the EPA's Mobile Source Observation Database (MSOD) to calculate emissions of CO and primary $PM_{2.5}$. Because tropospheric ozone and much $PM_{2.5}$ are created at a regional scale from locally emitted precursor pollutants, models such as Community Multiscale Air Quality (CMAQ) (Appel et al., 2007, 2008) must be employed to connect regional $PM_{2.5}$ and tropospheric ozone measurements to bioenergy-related precursor emissions. Emissions from liquid biofuel combustion in mobile sources can be estimated from country-scale estimates of consumption by fuel type combined with estimates of emissions from those fuels (Niven, 2005; Anderson, 2009; Gaffney and Marley, 2009). Emission estimates by fuel type should also be country-specific, as emissions vary with atmospheric conditions and policy-influenced design factors. For example, in some countries ethanol is consumed as an 85% blend with gasoline in specially equipped vehicles, whereas in other countries ethanol may be blended at lower levels with gasoline and consumed in all vehicles.

3.6. Indicator of productivity

One indicator, aboveground net primary productivity (ANPP), is recommended to assess the ecosystem productivity of bioenergy-associated land use (Table 2). The selection of this indicator is motivated by the importance of net primary productivity (NPP), which is defined as the net flux of carbon from the atmosphere into green plants per unit time and measures the rate of production of useful net energy by all plants in an ecosystem. NPP is a measure of the condition of both the land (e.g., soil fertility, topography, vegetation type, and prevailing weather conditions) and several ecological processes (including photosynthesis and autotrophic respiration as affected by local hydrology and temperature). Cramer et al. (1999) noted that "a better grasp upon the controls and distribution of . . . NPP . . . is pivotal for sustainable human use of the biosphere."

NPP manifests physically as total new plant biomass generated by photosynthesis per unit time (typically measured per year). Even so, the continual death and decay of plant tissue, especially belowground, as well as the import and export of organic compounds to and from the environment, make direct measurement of NPP difficult (Clark et al., 2001; Scurlock et al., 2002; Matamala et al., 2003).

Because of these and other challenges in directly measuring NPP, ANPP is often used as a substitute for NPP. Even measuring ANPP accurately is not trivial; however, certain difficult-to-measure components of ANPP (e.g., biomass consumed by herbivores or that dies and decomposes during the growing season) are often assumed to be small enough to ignore (Clark et al., 2001; Scurlock et al., 2002).

In agricultural systems, producers routinely measure yield, which in the case of biomass crops can serve as a proxy for ANPP. For some bioenergy systems in which not all aboveground biomass is harvested, such as corn starch ethanol, harvest indices are available for specific sites and systems (e.g., Pordesimo et al., 2004). A harvest index is the ratio of dry grain mass to total dry aboveground biomass for a given crop, and it varies somewhat with local varieties, conditions and management practices (Prince et al., 2001).

Because ANPP can be roughly approximated for both managed and unmanaged ecosystems, it provides a simple way to compare ecosystems that may differ dramatically in many respects. In cases where bioenergy feedstock crops replace less intensively managed ecosystems, the yield or estimated annual aboveground biomass of the feedstock crop can be compared to the ANPP of the prior ecosystem, measured either before bioenergy system implementation or on similar nearby proxy sites. Coupled with harvest indices to estimate NPP based on ANPP, such comparisons can also serve as one component for calculating the effects of land-use change on carbon dioxide flux.

4. Discussion

4.1. Developing and testing suite of indicators

These 19 indicators collectively represent how bioenergy systems may affect environmental sustainability with respect to soil quality, water quality and quantity, greenhouse gas concentrations, biodiversity, air quality, and productivity. Transitions from fossil-fuel based energy systems to bioenergy systems can affect environmental sustainability because of increases or decreases in various anthropogenic stresses, including resource exploitation; changes in land use, water use, and disturbance regime; and emissions of waste, pollutants, and greenhouse gases. Measured over time, this suite of indicators should reveal many of the effects of changes in these stressors not only pertaining to the current state of ecosystems but also relating to their resilience (Folke et al., 2004).

The suite of indicators presented here was selected with the goal of being useful in reflecting the environmental sustainability of a wide range of bioenergy systems. Even so, it is clear that particular applications may require modifications to the proposed suite of indicators as discussed in Section 4.2. The range of bioenergy systems includes variation in management and environmental context such as differences in feedstock choice, tillage and inputs, processing pathways, past land use, climate, and soil type. The desired utility of the suite of indicators across this range of systems includes the extent to which the indicators provide information as expected regarding environmental effects of concern as well as whether any indicators in the suite prove redundant with each other. It also includes the extent to which indicators are feasible, given available resources of money, time, access, and expertise. The success of this indicator suite at meeting these goals must be evaluated through field testing before it can be adopted.

Field testing consists of measuring the full suite of indicators in a set of established or pilot bioenergy systems. This set of systems should represent the range of potential production pathways and may require testing at various scales. One test with respect to feedstock production would consist of replicated pairs of experimental watersheds with each pair including a watershed that supports bioenergy production and a watershed that does not. Watersheds represent an ideal spatial resolution of focus for water quality and quantity indicators, which are most easily interpreted in the context of whole-watershed treatments.

In addition to assessing whether the suite meets goals relating to information and feasibility, field testing can also help in estimating variability and establishing appropriate targets for the suite of indicators in the context of particular bioenergy systems. By "variability" we mean the dispersion of an indicator's values both among the variety of bioenergy systems and within those with similar environmental and management context. Estimates of variability are needed to calculate the power of statistical tests performed to compare indicators over time, among different bioenergy systems, or between bioenergy systems and alternative land uses or energy sources.

Targets reflect knowledge about the sustainability of bioenergy systems given possible values of indicators and inform management responses to those values. Targets, along with guidelines for management actions, can be part of a comprehensive set of best management practices (BMPs) for bioenergy systems. Some targets take the form of thresholds or ranges, where measurements below, above, or between certain points are acceptable. Other targets might take the form of desired trends; for example, a target might be a continued increase in soil carbon over several years. Because the indicator suite presented here should be interpreted as an integrated whole, targets for each indicator depend on the overall effects of bioenergy systems on the environment as measured by the full suite of indicators, as well as on economic and social aspects of sustainability, as discussed in Section 4.4.

Finally, experience from field testing can also help in establishing detailed protocols for measuring the values of the indicators. In this paper we have provided references to standard methods for some indicators, but important details are left unspecified (e.g., frequency of measurement). Establishing more detailed protocols is an iterative process that should be part of field testing but should also extend into subsequent use of the suite of indicators. Standardization of protocols is desirable to increase comparability among indicator values estimated from different bioenergy systems. On the other hand, different situations require somewhat different methods, as discussed in Section 4.2.

The proposed indicator suite will undoubtedly be modified over time as knowledge and technology develop. As experience is gained with bioenergy systems and sustainability assessments, it will likely become apparent that some indicators measure attributes that are important but not changing with some bioenergy production pathways. And new indicators may prove necessary to measure conditions that change in unexpected ways. It may be useful to eliminate indicators in the former case and to add others in the second case in order to provide more detailed information about unexpected effects of bioenergy systems. In addition, advancements in technology will allow updates of the suite of environmental indicators for bioenergy sustainability. Ease of measurement is one reason that certain indicators have been chosen over others. More advanced and cost-effective instrumentation may allow for the replacement of some indicators identified here by others that measure related environmental effects more directly.

4.2. Adapting the suite of indicators for particular situations

The suite of 19 indicators presented here is not intended to be applied directly to particular bioenergy systems and management goals. Instead, this suite is intended as a basis or starting point for the selection of indicator suites for particular situations, which may require a subset or expansion of this proposed indicator suite. The choice of indicators for those suites may be driven by environmental context as well as cost. There are several advantages to giving special weight to a standard set of indicators when selecting indictor suites for specific purposes. First, to the extent that a standard suite has been field tested in a variety of conditions, stakeholders can have greater confidence in their suitability for similar scenarios. Second, if sets of indicators chosen for different applications are similar, their measured values are more likely to be comparable. Finally, improved coordination among those selecting indicators will improve coherence and efficiency in certification of sustainable biomass, avoid proliferation of redundant or nonaligned standards, and provide direction for the appropriate approach (van Dam et al., 2008).

The context of particular bioenergy systems and accompanying environmental concerns may suggest the selection of additional indicators beyond the 19 presented here. For example, indicators that measure contamination by heavy metals may be useful in systems where sewage sludge is used as fertilizer (McBride, 1995) or where bioenergy crops are expected to filter or immobilize contamination from other sources (e.g., Wu et al., 2003). Where genetically engineered feedstocks are grown, it may be important to monitor the spread of engineered genes and their effects on ecosystems (Snow et al., 2005). Similarly, where concern exists that feedstocks may become invasive in a given area (Barney and Ditomaso, 2008; Simberloff, 2008), their presence beyond the feedstock production site should be monitored. Where feedstock production is expected to exacerbate or ameliorate other biological invasions, it may be similarly important to monitor those invasive species on or near feedstock production sites. When water for irrigation is withdrawn from deep aquifers whose discharge to surface water is too slow or distant to be captured by base flow, groundwater levels should be monitored as an additional measure of consumptive water use.

By contrast, cost and management goals may require the elimination of some indicators. There are large costs involved in establishing a rigorous scientific monitoring of soil quality, water quality and quantity, greenhouse gases, biodiversity, air quality, and productivity. For example, although water indicators are important, they can be especially expensive to measure. Calculating flows, concentrations, and exports may require combinations of measurements using flumes or weirs, in situ instrumentation, and periodic sampling surveys, all in multiple locations and with high temporal resolution (Haan et al., 1994). The costs and feasibilities of measuring other indicators vary with different bioenergy systems. For example, the cost of accurately estimating net C_{eq} emissions varies depending on whether relevant data on fossil fuel consumption and feedstock management are readily available or must be collected specifically for indicator assessment. Similarly, the feasibility of estimating the abundance or habitat area of species of special concern depends on whether such species are already identified in a given system as well as the form and habit of those species.

In addition to adding or removing indicators to the suite, different situations and goals also require modifications to the protocols used in applying indicators. For example, measuring productivity in forests requires different techniques than measuring productivity in crops. In addition, cost constraints of efforts to estimate the suite of indicators may call for different methodologies relating to trade-offs between the cost, precision, and accuracy of specific protocols. Stakeholder goals may affect protocols as well. For example, bioenergy systems are often envisioned as integral parts of sustainable landscape designs (Dale et al., 2010a). Consideration of landscape patterns and diversity in planning feedstock production systems may result in environmental benefits such as increased biodiversity and decreased erosion and runoff pollution (Firbank, 2008; Dale et al., 2010b). To assess the success of management practices that consider landscape design, indicators might best be applied to extents larger than individual bioenergy operations.

4.3. Interpreting the suite of indicator measurements

Indicators should be interpreted in view of baseline conditions and the particular context of a proposed bioenergy system. Baseline conditions are a set of observations or data that are used for comparison to new activities or for a reference case. With regard to the environmental sustainability of bioenergy, baseline conditions attempt to characterize environmentally relevant aspects of a situation in which a given bioenergy system had not been implemented. Ideally, a comparison between indicator values and baseline conditions should reveal the marginal environmental effects of a bioenergy system. Some baseline conditions can be represented by initial values of indicators if measurements are taken before bioenergy operations are initiated. For example, indicators that characterize land-use attributes, such as those relating to soil and water, can be measured prior to bioenergy-related land-use change. As a proxy, when initial values of indicators are not available, baseline conditions can be measured in areas that are similar to the prior state of production land – most often at a nearby location that has similar weather, topography, soils, vegetation, drainage area/hydrology, and management practices as the initial conditions of the bioenergy production site. Similarly, air quality indicators, especially important in relation to processing facilities such as biorefineries, can be measured before the facility is brought on line or at a suitable proxy site; however, the complex regional dynamics of air pollutants such as ozone and $PM_{2.5}$ may complicate the selection of such sites.

Because business-as-usual scenarios for energy are based on fossil fuels, the baseline for bioenergy sustainability should consider environmental implications of fossil fuel exploration, drilling, mining, production, transportation, and use (Gorissen et al., 2010). However, data are rarely available to determine the full environmental effects of fossil fuel systems. Even so, life cycle assessment (LCA) for fossil fuel systems demonstrates that the environmental effects of those systems vary widely with geography and other factors (Furuholt, 1995).

In addition to baseline conditions, contextual variables must be used to interpret indicator measurements. Contextual variables measure characteristics of the operation of a bioenergy system that may affect the value of an indicator. Some contextual variables change with time but are beyond the direct control of operation managers. As an example, information on rainfall intensity and frequency is used to interpret measures of stream flow. Similarly, soil, water, and biodiversity indicators depend on disturbance regimes including the frequency and intensity of fire and floods. Some contextual variables are site characteristics that change little or not at all over time (e.g., land-use history, soil texture, slope, and aspect) and thus may be measured with lower frequency. Other contextual variables are aspects of land management, such as crop choice, tillage intensity, frequency of burning, percentage of residue removed, and applications of fertilizers, pesticides, and herbicides. For example, measures of soil nitrogen and stream nitrate should be considered in the context of the amount of nitrogen fertilizer applied to the soil. These management-related contextual variables can further be divided into those under the control of bioenergy operation managers and those under the control of other resource managers, such as farmers growing non-bioenergy crops upstream from bioenergy crops. Those variables under direct control of bioenergy operation managers serve not only as contextual variables but also as objects of manipulation for the application of BMPs. Table 2 lists examples of management-related contextual variables with respect to each of the indicators presented.

As an indicator of environmental sustainability, measurement of aboveground net primary productivity (ANPP) is especially important to interpret along with contextual variables. For example, rainfall records may allow a decline in feedstock ANPP to be attributed to unsustainable soil degradation or to drought or other conditions beyond the control of land managers. Similarly, increasing ANPP may reflect increasing sustainability if accompanied by the adoption of precision agriculture techniques or by a shift to crops or crop varieties better suited for a given site. On the other hand, such an increasing trend may reflect decreasing sustainability if accompanied by increases in fertilizer or irrigation input. As a third example, the maintenance of ANPP at relatively consistent levels in the context of disturbances such as hurricane, drought, or disease may reflect a resilient agroecosystem.

In response to given management practices, some indicators are likely to change in favorable directions and others in unfavorable directions. Such differences represent the unavoidable tradeoffs that make sustainable management challenging. To some extent, determining optimal management practice depends on inherently subjective judgments on the part of stakeholders regarding the importance of different indicators or the extent that options for potential environmental benefits should be maintained over time. A multivariate analysis of the 19 indicators' values will provide a basis for stakeholders to discuss characteristics of environmentally sustainable bioenergy systems. Sustainability polygons (also known as cobweb polygons, star plots, or radar charts) represent one method for visualizing the measured values of suites of indicators as multivariate observations (e.g., Gomez et al., 1996; de Vries et al., 2010).

4.4. Economic and social sustainability

Indicators of environmental sustainability also provide information about economic and social sustainability, because economies and societies rely on the continued provision of ecosystem services, defined as the benefits people obtain from ecosystems (MEA, 2005). The indicators of environmental sustainability identified here relate to the provisioning, regulating, cultural, and supporting ecosystem services (MEA, 2005) that can be enhanced or degraded by bioenergy systems. However, because sustainable economies and societies rely on conditions other than the provision of ecosystem services, indicators of social and economic sustainability are needed in addition to the indicators of environmental sustainability proposed in this paper (Niemi and McDonald, 2004). Developing comprehensive suites of sustainability indicators for bioenergy is the goal of the Roundtable on Sustainable Biofuels (RSB, 2010), the Global Bioenergy Partnership (GBEP, 2010), and other national and international organizations. The current paper strives to support those efforts by presenting a short list of environmental indicators that can be used to evaluate bioenergy systems.

5. Conclusion

We identify a suite of 19 indicators in six categories to measure the environmental sustainability of bioenergy systems. The suite is intended to be a practical toolset for capturing key environmental effects of bioenergy across a range of bioenergy systems, including different pathways, locations, and management practices. To evaluate the hypothesis that the suite meets this goal, and also to help measure variability and establish appropriate targets, the suite should be field tested in systems spanning a wide variety of conditions. If the hypothesis is confirmed, the suite can be implemented more broadly, modified as necessary for particular contexts. This broader implementation will further two goals. First, it will help stakeholders judge the relative environmental sustainability of different bioenergy systems, including the question of which feedstocks, management practices, and post-production processes are appropriate for different locations as well as the question of how bioenergy systems compare with alternative energy systems. Second, it will help provide an empirical foundation for indicators designed to assess environmental sustainability based on the predicted effects of management practices, such as many of the indicators proposed for use in certifying sustainable bioenergy systems (e.g., GBEP, 2010; RSB, 2010).

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References

Adair, C.E., Reich, P., Hobbie, S., Knops, J., 2009. Interactive effects of time, CO_2, N, and diversity on total belowground carbon allocation and ecosystem carbon storage in a grassland community. Ecosystems 12, 1037–1052.
Adler, P.R., Del Grosso, S.J., Parton, W.J., 2007. Life-cycle assessment of net greenhouse-gas flux for bioenergy cropping systems. Ecol. Appl. 17, 675–691.
AIRNow, 2010. AIRNow. http://www.airnow.gov/ (accessed January 2011).
Alexander, R.B., Smith, R.A., Schwarz, G.E., Boyer, E.W., Nolan, J.V., Brakebill, J.W., 2008. Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River basin. Environ. Sci. Technol. 42, 822–830.
Anderson, L.G., 2009. Ethanol fuel use in Brazil: air quality impacts. Energy Environ. Sci. 2, 1015–1037.
Aneja, V.P., Schlesinger, W.H., Erisman, J.W., 2009. Effects of agriculture upon the air quality and climate: research, policy, and regulations. Environ. Sci. Technol. 43, 4234–4240.
Appel, K.W., Gilliland, A.B., Sarwar, G., Gilliam, R.C., 2007. Evaluation of the Community Multiscale Air Quality (CMAQ) model version 4.5: sensitivities impacting model performance. Part I. Ozone. Atmos. Environ. 41, 9603–9615.
Appel, K.W., Bhave, P.V., Gilliland, A.B., Sarwar, G., Roselle, S.J., 2008. Evaluation of the Community Multiscale Air Quality (CMAQ) model version 4.5: sensitivities impacting model performance. Part II. Particulate matter. Atmos. Environ. 42, 6057–6066.
Atkinson, R., 2000. Atmospheric chemistry of VOCs and NO*x*. Atmos. Environ. 34, 2063–2101.
Bangert, R.K., Slobodchikoff, C.N., 2006. Conservation of prairie dog ecosystem engineering may support arthropod beta and gamma diversity. J. Arid Environ. 67, 100–115.
Barney, J.N., Ditomaso, J.M., 2008. Nonnative species and bioenergy: are we cultivating the next invader? Bioscience 58, 64–70.
Berndes, G., 2002. Bioenergy and water—the implications of large-scale bioenergy production for water use and supply. Global Environ. Change 12, 253–271.
Biala, K., Peeters, A., Muys, B., Hermy, M., Brouckaert, V., García, V., Van der Veken, B., Valckx, J., 2005. Biodiversity indicators as a tool to assess sustainability levels of agro-ecosystems, with a special consideration of grassland areas. Opt. Méditerran., Ser. A 67, 439–443.
Bouwman, A.F., van Grinsven, J.J.M., Eickhout, B., 2010. Consequences of the cultivation of energy crops for the global nitrogen cycle. Ecol. Appl. 20, 101–109.
Bremner, J.M., Mulvaney, C.S., 1982. Nitrogen: total. In: Page, A.L., Miller, R.H., Keeney, D.R. (Eds.), Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties. , 2nd ed. American Society of Agronomy, Soil Science Society of America Inc., Madison, WI.
Brunekreef, B., Forsberg, B., 2005. Epidemiological evidence of effects of coarse airborne particles on health. Eur. Respir. J. 26, 309–318.
Bruntland, G.H. (Ed.), 1987. Our Common Future: The World Commission on Environment and Development. Oxford University Press, Oxford.

Buchanan, T.J., Somers, W.P., 1969. Discharge measurements at gaging stations. Techniques of Water-Resources Investigations, vol. 3. U.S. Geological Survey.
Buchholz, T., Luzadis, V.A., Volk, T.A., 2009. Sustainability criteria for bioenergy systems: results from an expert survey. J. Cleaner Prod. 17, S86–S98.
Bunn, S.E., Arthington, A.H., 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environ. Manage. 30, 492–507.
Chen, T.M., Gokhale, J., Shofer, S., Kuschner, W.G., 2007. Outdoor air pollution: nitrogen dioxide, sulfur dioxide, and carbon monoxide health effects. Am. J. Med. Sci. 333, 249–256.
Cherubini, F., 2010. GHG balances of bioenergy systems—overview of key steps in the production chain and methodological concerns. Renew. Energy 35, 1565–1573.
Cherubini, F., Bird, N.D., Cowie, A., Jungmeier, G., Schlamadinger, B., Woess-Gallasch, S., 2009. Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: key issues, ranges and recommendations. Resour. Conserv. Recycl. 53, 434–447.
Clark, D.A., Brown, S., Kicklighter, D.W., Chambers, J.Q., Thomlinson, J.R., Ni, J., 2001. Measuring net primary production in forests: concepts and field methods. Ecol. Appl. 11, 356–370.
Cramer, W., Kicklighter, D.W., Bondeau, A., Iii, B.M., Churkina, G., Nemry, B., Ruimy, A., Schloss, A.L., 1999. The participants of the Potsdam NPP model intercomparison. Comparing global models of terrestrial net primary productivity (NPP): overview and key results. Global Change Biol. 5 (S1), 1–15.
Crutzen, P.J., Mosier, A.R., Smith, K.A., Winiwarter, W., 2008. N_2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. Atmos. Chem. Phys. 8, 389–395.
CSBP, 2010. Draft Provisional Standard for Sustainable Production of Agricultural Biomass. Council on Sustainable Biomass Production, http://www.csbp.org/files/survey/CSBP_Provisional_Standard.pdf (accessed January 2011).
Dalal, R.C., Wang, W.J., Robertson, G.P., Parton, W.J., 2003. Nitrous oxide emission from Australian agricultural lands and mitigation options: a review. Aust. J. Soil Res. 41, 165–195.
Dale, V.H., Beyeler, S.C., 2001. Challenges in the development and use of ecological indicators. Ecol. Indic. 1, 3–10.
Dale, V.H., Kline, K.L., Weins, J., Fargione, J., 2010a. Biofuels: implications for land use and biodiversity. In: Biofuels and Sustainability Reports. Ecological Society of America, http://www.esa.org/biofuelsreports/files/ESA Biofuels Report_VH Dale et al.pdf (accessed January 2011).
Dale, V.H., Lowrance, R., Mulholland, P.J., Robertson, G.P., 2010b. Bioenergy sustainability at the regional scale. Ecol. Soc. 15, http://www.ecologyandsociety.org/vol15/iss4/art23/ (accessed January 2011).
Dale, V.H., Polasky, S., 2007. Measures of the effects of agricultural practices on ecosystem services. Ecol. Econ. 64, 286–296.
Davis, S.C., Anderson-Teixeira, K.J., DeLucia, E.H., 2009. Life-cycle analysis and the ecology of biofuels. Trends Plant Sci. 14, 140–146.
de Fraiture, C., Giordano, M., Liao, Y.S., 2008. Biofuels and implications for agricultural water use: blue impacts of green energy. Water Policy 10, 67–81.
de Lima, J., Singh, V.P., de Lima, M.I.P., 2003. The influence of storm movement on water erosion: storm direction and velocity effects. Catena 52, 39–56.
de Vries, S.C., van de Ven, G.W.J., van Ittersum, M.K., Giller, K.E., 2010. Resource use efficiency and environmental performance of nine major biofuel crops, processed by first-generation conversion techniques. Biomass Bioenergy 34, 588–601.
Del Grosso, S.J., Ogle, S.M., Parton, W.J., Breidt, F.J., 2010. Estimating uncertainty in N_2O emissions from US cropland soils. Global Biogeochem. Cycles 24 (GB1009), 1–12.
Doran, J.W., Jones, A.J. (Eds.), 1996. Methods for Assessing Soil Quality. Soil Science Society of America, Inc., Madison, WI.
Doran, J.W., Parkin, T.B., 1996. Quantitative indicators of soil quality: a minimum data set. In: Doran, J.W., Jones, A.J. (Eds.), Methods for Assessing Soil Quality. Soil Science Society of America, Inc., Madison, WI, pp. 25–37.
Earles, R., Williams, P., 2005. Sustainable Agriculture: An Introduction. National Center for Appropriate Technology, http://attra.ncat.org/attra-pub/PDF/sustagintro.pdf (accessed January 2011).
Eaton, A.D., Clesceri, L.S., Rice, E.W., Greenberg, A.E., Franson, M.A.H. (Eds.), 2005. Standard Methods for the Examination of Water and Wastewater. , 21st ed. American Public Health Association, Washington, D.C..
Elser, J.J., Bracken, M.E.S., Cleland, E.E., Gruner, D.S., Harpole, W.S., Hillebrand, H., Ngai, J.T., Seabloom, E.W., Shurin, J.B., Smith, J.E., 2007. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecol. Lett. 10, 1135–1142.
Energy Independence and Security Act of 2007. 110-140.2007. http://frwebgate.access.gpo.gov/cgi-bin/getdoc.cgi?dbname=110_cong_public_laws&docid=f:publ140.110.pdf (accessed January 2011).
EPA, 2010. National Ambient Air Quality Standards (NAAQS). United States Environmental Protection Agency, http://epa.gov/air/criteria.html (accessed January 2011).
Eugster, W., Zeyer, K., Zeeman, M., Michna, P., Zingg, A., Buchmann, N., Emmenegger, L., 2007. Methodical study of nitrous oxide eddy covariance measurements using quantum cascade laser spectrometery over a Swiss forest. Biogeosciences 4, 927–939.
European Parliament and Council, 2009. Directive 2009/28/EC of the European Parliament and of the Council. Official J. Eur. Union 52 (L140), 16–62, http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2009:140:0016:0062:en:PDF (accessed January 2011).
Farquharson, R., Baldock, J., 2008. Concepts in modelling N_2O emissions from land use. Plant Soil 309, 147–167.
Firbank, L.G., 2008. Assessing the ecological impacts of bioenergy projects. Bioenergy Res. 1, 12–19.
Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., Holling, C.S., 2004. Regime shifts, resilience, and biodiversity in ecosystem management. Annu. Rev. Ecol., Evol. Syst. 35, 557–581.
Francoeur, S.N., 2001. Meta-analysis of lotic nutrient amendment experiments: detecting and quantifying subtle responses. J. N. Am. Benthol. Soc. 20, 358–368.
Furuholt, E., 1995. Life cycle assessment of gasoline and diesel. Resour. Conserv. Recycl. 14, 251–263.
Gaffney, J.S., Marley, N.A., 2009. The impacts of combustion emissions on air quality and climate—from coal to biofuels and beyond. Atmos. Environ. 43, 23–36.
Gardiner, M.A., Tuell, J.K., Isaacs, R., Gibbs, J., Ascher, J.S., Landis, D.A., 2010. Implications of three biofuel crops for beneficial arthropods in agricultural landscapes. Bioenergy Res. 3, 6–19.
Garten, C.T., Ashwood, T.L., Dale, V.H., 2003. Effect of military training on indicators of soil quality at Fort Benning, Georgia. Ecol. Indic. 3, 171–179.
Garten, C.T., Smith, J.L., Tyler, D.D., Amonette, J.E., Bailey, V.L., Brice, D.J., Castro, H.F., Graham, R.L., Gunderson, C.A., Izaurralde, R.C., Jardine, P.M., Jastrow, J.D., Kerley, M.K., Matamala, R., Mayes, M.A., Metting, F.B., Miller, R.M., Moran, K.K., Post, W.M., Sands, R.D., Schadt, C.W., Phillips, J.R., Thomson, A.M., Vugteveen, T., West, T.O., Wullschleger, S.D., 2010. Intra-annual changes in biomass, carbon, and nitrogen dynamics at 4-year old switchgrass field trials in west Tennessee, USA. Agric. Ecosyst. Environ. 136, 177–184.
GBEP, 2010. Second Draft of GBEP Sustainability Criteria and Indicators for Bioenergy. Global Bioenergy Partnership, http://www.globalbioenergy.org/fileadmin/user_upload/gbep/docs/partners_only/sust_docs/2nd_DRAFT_of_GBEP_Criteria_Indicators_with_TEMPLATES.doc (accessed January 2011).
Gnansounou, E., Dauriat, A., Villegas, J., Panichelli, L., 2009. Life cycle assessment of biofuels: energy and greenhouse gas balances. Bioresour. Technol. 100, 4919–4930.
Gomez, A.A., Swete Kelly, D.E., Seyers, J.K., Coughlan, K.J., 1996. Measuring sustainability of agricultural systems at the farm level. In: Doran, J.W., Jones, A.J. (Eds.), Methods for Assessing Soil Quality. Soil Science Society of America, Inc., Madison, WI, pp. 401–410.
Gorissen, L., Buytaert, V., Cuypers, D., Dauwe, T., Pelkmans, L., 2010. Why the debate about land use change should not only focus on biofuels. Environ. Sci. Technol. 44, 4046–4049.
Haan, C.T., Barfield, B.J., Hayes, J.C., 1994. Design Hydrology and Sedimentology for Small Catchments. Academic Press, San Diego.
Hansen, J., Sato, M., Ruedy, R., Lo, K., Lea, D.W., Medina-Elizade, M., 2006. Global temperature change. Proc. Natl. Acad. Sci. U.S.A. 103, 14288–14293.
Hatchell, G.E., Ralston, C.W., Foil, R.R., 1970. Soil disturbances in logging: effects on soil characteristics and growth of loblolly pine in the Atlantic Coastal Plain. J. For. 68, 772–775.
Hecht, A.D., Shaw, D., Bruins, R., Dale, V., Kline, K., Chen, A., 2009. Good policy follows good science: using criteria and indicators for assessing sustainable biofuel production. Ecotoxicology 18, 1–4.
Heink, U., Kowarik, I., 2010. What are indicators? On the definition of indicators in ecology and environmental planning. Ecol. Indic. 10, 584–593.
Hellebrand, H.J., Scholz, V., Kern, J., 2008. Fertiliser induced nitrous oxide emissions during energy crop cultivation on loamy sand soils. Atmos. Environ. 42, 8403–8411.
Hudson, N.W., 1993. Field Measurement of Soil Erosion and Runoff. Food and Agriculture Organization of the United Nations, Rome, http://www.fao.org/docrep/t0848e/t0848e00.htm (accessed January 2011).
Huffaker, R., 2010. Protecting water resources in biofuels production. Water Policy 12, 129–134.
IPCC, 2007. In: Core Writing Team, Pachauri, R.K., Reisinger, A. (Eds.), Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switzerland, http://www.ipcc.ch/publications_and_data/publications_ipcc_fourth_assessment_report_synthesis_report.htm (accessed January 2011).
Ito, A., Penner, J.E., 2004. Global estimates of biomass burning emissions based on satellite imagery for the year 2000. J. Geophys. Res. 109 (D14S05), 1–18.
Jordan, N., Boody, G., Broussard, W., Glover, J.D., Keeney, D., McCown, B.H., McIsaac, G., Muller, M., Murray, H., Neal, J., Pansing, C., Turner, R.E., Warner, K., Wyse, D., 2007. Environment: sustainable development of the agricultural bio-economy. Science 316, 1570–1571.
Karlen, D.L., Ditzler, C.A., Andrews, S.S., 2003. Soil quality: why and how? Geoderma 114, 145–156.
Keeney, D., 2008. Ethanol USA. Environ. Sci. Technol 43, 8–11.
Kellogg, R.L., Nehring, R., Grube, A., Goss, D.W., Plotkin, S., 2000. Environmental indicators of pesticide leaching and runoff from farm fields. In: Paper presented at: Agricultural Productivity: Data, Methods, and Measures , March 9–10, 2000, Washington, D.C., http://www.nrcs.usda.gov/technical/NRI/pubs/eip_pap.html (accessed January 2011).
Kiely, T., Donaldson, D., Grube, A., 2004. Pesticide Industry Sales and Usage: 2000 and 2001 Market Estimates. United States Environmental Protection Agency, http://www.epa.gov/opp00001/pestsales/01pestsales/market_estimates2001.pdf (accessed January 2011).
Kline, K., Dale, V.H., Lee, R., Leiby, P., 2009. In defense of biofuels, done right. Issues Sci. Technol. 25, 75–84.

Laden, F., Neas, L.M., Dockery, D.W., Schwartz, J., 2000. Association of fine particulate matter from different sources with daily mortality in six US cities. Environ. Health Perspect. 108, 941–947.
Lawler, D.M., Petts, G.E., Foster, I.D.L., Harper, S., 2006. Turbidity dynamics during spring storm events in an urban headwater river system: the Upper Tame, West Midlands, UK. Sci. Total Environ. 360, 109–126.
Lawler, J.J., White, D., Sifneos, J.C., Master, L.L., 2003. Rare species and the use of indicator groups for conservation planning. Conserv. Biol. 17, 875–882.
Liska, A.J., Cassman, K.G., 2008. Towards standardization of life-cycle metrics for biofuels: greenhouse gas emissions mitigation and net energy yield. J. Biobased Mater. Bioenergy 2, 187–203.
Macedo, I.C., Seabra, J.E.A., Silva, J.E.A.R., 2008. Green house gases emissions in the production and use of ethanol from sugarcane in Brazil: the 2005/2006 averages and a prediction for 2020. Biomass Bioenergy 32, 582–595.
Malm, W., 1999. Introduction to Visibility. Cooperative Institute for Research in the Atmosphere, Colorado State University, Fort Collins, CO, http://www.epa.gov/visibility/pdfs/introvis.pdf (accessed January 2011).
Mann, L., Tolbert, V., 2000. Soil sustainability in renewable biomass plantings. Ambio 29, 492–498.
Master, L.L., 1991. Assessing threats and setting priorities for conservation. Conserv. Biol. 5, 559–563.
Matamala, R., Gonzalez-Meler, M.A., Jastrow, J.D., Norby, R.J., Schlesinger, W.H., 2003. Impacts of fine root turnover on forest NPP and soil C sequestration potential. Science 302, 1385–1387.
McBride, M.B., 1995. Toxic metal accumulation from agricultural use of sludge: are USEPA regulations protective? J. Environ. Qual. 24, 5–18.
McCoy, E.D., Mushinsky, H.R., 2007. Estimates of minimum patch size depend on the method of estimation and the condition of the habitat. Ecology 88, 1401–1407.
McMichael, A.J., Woodruff, R.E., Hales, S., 2006. Climate change and human health: present and future risks. Lancet 367, 859–869.
MEA, 2005. Ecosystems and Human Well-being: Synthesis. Island Press, Washington, D.C., http://www.maweb.org/documents/document.356.aspx.pdf (accessed January 2011).
Mehlich, A., 1984. Mehlich 3 soil test extractant: a modification of Mehlich 2 extractant. Commun. Soil Sci. Plant Anal. 15, 1409–1416.
Monti, A., Venturi, P., Elbersen, H.W., 2001. Evaluation of the establishment of lowland and upland switchgrass (*Panicum virgatum* L.) varieties under different tillage and seedbed conditions in northern Italy. Soil Tillage Res. 63, 75–83.
Moscatelli, M.C., Lagomarsino, A., Marinari, S., De Angelis, P., Grego, S., 2005. Soil microbial indices as bioindicators of environmental changes in a poplar plantation. Ecol. Indic. 5, 171–179.
Murray, L.D., Best, L.B., Jacobsen, T.J., Braster, M.L., 2003. Potential effects on grassland birds of converting marginal cropland to switchgrass biomass production. Biomass Bioenergy 25, 167–175.
Neftel, A., Flechard, C., Ammann, C., Conen, F., Emmenegger, L., Zeyer, K., 2007. Experimental assessment of N_2O background fluxes in grassland systems. Tellus Ser. B: Chem. Phys. Meteorol. 59, 470–482.
Niemi, G.J., McDonald, M.E., 2004. Application of ecological indicators. Annu. Rev. Ecol., Evol. Syst. 35, 89–111.
Niven, R.K., 2005. Ethanol in gasoline: environmental impacts and sustainability review article. Renew. Sustain. Energy Rev. 9, 535–555.
NRC, 1977. Ozone and Other Photochemical Oxidants. Printing and Publishing Office, National Academy of Sciences, Washington, D.C.
NRC, 2008a. Monitoring Climate Change Impacts: Metrics at the Intersection of the Human and Earth Systems. The National Academies Press, Washington, D.C., http://www.nap.edu/catalog.php?record_id=12965 (accessed January 2011).
NRC, 2008b. Water Implications of Biofuels Production in the United States. The National Academies Press, Washington, D.C., http://www.nap.edu/catalog.php?record_id=12039 (accessed January 2011).
NRCS, 2009. Total Organic Carbon. USDA Natural Resources Conservation Service, http://soils.usda.gov/sqi/assessment/files/toc_sq_biological_indicator_sheet.pdf (accessed January 2011).
Olsen, S.R., Cole, C.V., Watanabe, F.S., Dean, L.A., 1954. Estimation of available phosphorus in soils by extraction with sodium bicarbonate. In: USDA Circular 939. U.S. Government Printing Office, Washington, D.C.
Parton, W.J., Hartman, M., Ojima, D., Schimel, D., 1998. DAYCENT and its land surface submodel: description and testing. Global Planet. Change 19, 35–48.
Pattison, A.B., Moody, P.W., Badcock, K.A., Smith, L.J., Armour, J.A., Rasiah, V., Cobon, J.A., Gulino, L.M., Mayer, R., 2008. Development of key soil health indicators for the Australian banana industry. Appl. Soil Ecol. 40, 155–164.
Pope, C.A., Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K., Thurston, G.D., 2002. Lung cancer, cardiopulmonary mortality, and long-term exposure to fine particulate air pollution. J. Am. Med. Assoc. 287, 1132–1141.
Pordesimo, L.O., Edens, W.C., Sokhansanj, S., 2004. Distribution of aboveground biomass in corn stover. Biomass Bioenergy 26, 337–343.
Power, M.E., Tilman, D., Estes, J.A., Menge, B.A., Bond, W.J., Mills, L.S., Daily, G., Castilla, J.C., Lubchenco, J., Paine, R.T., 1996. Challenges in the quest for keystones. Bioscience 46, 609–620.
Prince, S.D., Haskett, J., Steininger, M., Strand, H., Wright, R., 2001. Net primary production of U.S. Midwest croplands from agricultural harvest yield data. Ecol. Appl. 11, 1194–1205.
Reeves, D.W., 1997. The role of soil organic matter in maintaining soil quality in continuous cropping systems. Soil Tillage Res. 43, 131–167.
Robertson, G.P., Dale, V.H., Doering, O.C., Hamburg, S.P., Melillo, J.M., Wander, M.M., Parton, W.J., Adler, P.R., Barney, J.N., Cruse, R.M., Duke, C.S., Fearnside, P.M., Follett, R.F., Gibbs, H.K., Goldemberg, J., Mladenoff, D.J., Ojima, D., Palmer, M.W., Sharpley, A., Wallace, L., Weathers, K.C., Wiens, J.A., Wilhelm, W.W., 2008. Agriculture: sustainable biofuels redux. Science 322, 49–50.
RSB, 2010. RSB Principles and Criteria. École Polytechnique Fédérale de Lausanne, http://rsb.epfl.ch/files/content/sites/rsb2/files/Biofuels/Version 2/PCs V2/10-11-12 RSB PCs Version 2.pdf (accessed January 2011).
Schäfer, R.B., Caquet, T., Siimes, K., Mueller, R., Lagadic, L., Liess, M., 2007. Effects of pesticides on community structure and ecosystem functions in agricultural streams of three biogeographical regions in Europe. Sci. Total Environ. 382, 272–285.
Scharlemann, J.P.W., Laurance, W.F., 2008. Environmental science: how green are biofuels? Science 319, 43–44.
Scurlock, J.M.O., Johnson, K., Olson, R.J., 2002. Estimating net primary productivity from grassland biomass dynamics measurements. Global Change Biol. 8, 736–753.
Seinfeld, J.H., Pankow, J.F., 2003. Organic atmospheric particulate material. Annu. Rev. Phys. Chem. 54, 121–140.
Semere, T., Slater, F.M., 2007. Ground flora, small mammal and bird species diversity in miscanthus (*Miscanthus* × *giganteus*) and reed canary-grass (*Phalaris arundinacea*) fields. Biomass Bioenergy 31, 20–29.
Sharpley, A.N., Kleinman, P.J.A., Heathwaite, A.L., Gburek, W.J., Folmar, G.J., Schmidt, J.P., 2008. Phosphorus loss from an agricultural watershed as a function of storm size. J. Environ. Qual. 37, 362–368.
Shipley, B.K., Reading, R.P., 2006. A comparison of herpetofauna and small mammal diversity on black-tailed prairie dog (Cynomys ludovicianus) colonies and non-colonized grasslands in Colorado. J. Arid Environ. 66, 27–41.
Shurpali, N.J., Strandman, H., Kilpelainen, A., Huttunen, J., Hyvonen, N., Biasi, C., Kellomaki, S., Martikainen, P.J., 2010. Atmospheric impact of bioenergy based on perennial crop (reed canary grass, *Phalaris arundinaceae* L.) cultivation on a drained boreal organic soil. Global Change Biol. Bioenergy 2, 130–138.
Simberloff, D., 2008. Invasion biologists and the biofuels boom: cassandras or colleagues? Weed Sci. 56, 867–872.
Smeets, E.M.W., Faaij, A.P.C., 2010. The impact of sustainability criteria on the costs and potentials of bioenergy production—applied for case studies in Brazil and Ukraine. Biomass Bioenergy 34, 319–333.
Smith, V.H., Tilman, G.D., Nekola, J.C., 1999. Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. Environ. Pollut. 100, 179–196.
Snow, A.A., Andow, D.A., Gepts, P., Hallerman, E.M., Power, A., Tiedje, J.M., Wolfenbarger, L.L., 2005. Genetically engineered organisms and the environment: current status and recommendations. Ecol. Appl. 15, 377–404.
Snyder, C.S., Bruulsema, T.W., Jensen, T.L., Fixen, P.E., 2009. Review of greenhouse gas emissions from crop production systems and fertilizer management effects. Agric. Ecosyst. Environ. 133, 247–266.
Stanciulescu, V., Fleming, J.S., 2006. Life cycle assessment of transportation fuels and GHGenius. In: EIC Climate Change Technology, 2006 IEEE , pp. 1–11.
Stone, K.C., Hunt, P.G., Cantrell, K.B., Ro, K.S., 2010. The potential impacts of biomass feedstock production on water resource availability. Bioresour. Technol. 101, 2014–2025.
Sullivan, P., 2003. Applying the Principles of Sustainable Farming. National Center for Appropriate Technology, http://attra.ncat.org/attra-pub/PDF/Transition.pdf (accessed January 2011).
Tolbert, V.R., Todd, D.E., Mann, L.K., Jawdy, C.M., Mays, D.A., Malik, R., Bandaranayake, W., Houston, A., Tyler, D., Pettry, D.E., 2002. Changes in soil quality and below-ground carbon storage with conversion of traditional agricultural crop lands to bioenergy crop production. Environ. Pollut. 116, S97–S106.
Townsend, C.L., Maynard, R.L., 2002. Effects on health of prolonged exposure to low concentrations of carbon monoxide. Occup. Environ. Med. 59, 708–711.
Turlure, C., Choutt, J., Van Dyck, H., Baguette, M., Schtickzelle, N., 2010. Functional habitat area as a reliable proxy for population size: case study using two butterfly species of conservation concern. J. Insect Conserv. 14, 379–388.
Unger, P.W., Kaspar, T.C., 1994. Soil compaction and root growth: a review. Agron. J. 86, 759–766.
Ussiri, D.A.N., Lal, R., Jarecki, M.K., 2009. Nitrous oxide and methane emissions from long-term tillage under a continuous corn cropping system in Ohio. Soil Tillage Res. 104, 247–255.
van Dam, J., Junginger, M., Faaij, A., Jurgens, I., Best, G., Fritsche, U., 2008. Overview of recent developments in sustainable biomass certification. Biomass Bioenergy 32, 749–780.
Venkataraman, C., Rao, G.U.M., 2001. Emission factors of carbon monoxide and size-resolved aerosols from biofuel combustion. Environ. Sci. Technol. 35, 2100–2107.
Vlassak, K., 1970. Total soil nitrogen and nitrogen mineralization. Plant Soil 32, 27–32.
Wang, M., 2002. Fuel choices for fuel-cell vehicles: well-to-wheels energy and emission impacts. J. Power Sources 112, 307–321.
Wang, M., Wu, M., Huo, H., Liu, J.H., 2008. Life-cycle energy use and greenhouse gas emission implications of Brazilian sugarcane ethanol simulated with the GREET model. Int. Sugar J. 110, 527–545.
Ward, M.H., deKok, T.M., Levallois, P., Brender, J., Gulis, G., Nolan, B.T., VanDerslice, J., 2005. Workgroup report: drinking-water nitrate and health—recent findings and research needs. Environ. Health Perspect. 113, 1607–1614.
West, T.O., Brandt, C.C., Baskaran, L.M., Hellwinckel, C.M., Mueller, R., Bernacchi, C.J., Bandaru, V., Yang, B., Wilson, B.S., Marland, G., Nelson, R.G., Ugarte, D.G.D., Post, W.M., 2010. Cropland carbon fluxes in the United States: increasing geospatial resolution of inventory-based carbon accounting. Ecol. Appl. 20, 1074–1086.

Williams, P.R.D., Inman, D., Aden, A., Heath, G.A., 2009. Environmental and sustainability factors associated with next-generation biofuels in the U.S.: what do we really know? Environ. Sci. Technol. 43, 4763–4775.

Wood, P.J., Armitage, P.D., 1997. Biological effects of fine sediment in the lotic environment. Environ. Manage. 21, 203–217.

Wu, J., Mersie, W., Atalay, A., Seybold, C.A., 2003. Copper retention from runoff by switchgrass and tall fescue filter strips. J. Soil Water Conserv. 58, 67–73.

Wu, M., Mintz, M., Wang, M., Arora, S., 2009. Water consumption in the production of ethanol and petroleum gasoline. Environ. Manage. 44, 981–997.

Yevich, R., Logan, J.A., 2003. An assessment of biofuel use and burning of agricultural waste in the developing world. Global Biogeochem. Cycles 17 (1095), 1–40.