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Comparing Scales of Environmental Effects from Gasoline and Ethanol Production

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Abstract Understanding the environmental effects of alternative fuel production is critical to characterizing the sustainability of energy resources to inform policy and regulatory decisions. The magnitudes of these environmental effects vary according to the intensity and scale of fuel production along each step of the supply chain. We compare the spatial extent and temporal duration of ethanol and gasoline production processes and environmental effects based on a literature review and then synthesize the scale differences on space–time diagrams. Comprehensive assessment of any fuel-production system is a moving target, and our analysis shows that decisions regarding the selection of spatial and temporal boundaries of analysis have tremendous influences on the comparisons. Effects that strongly differentiate gasoline and ethanol-supply chains in terms of scale are associated with when and where energy resources are formed and how they are extracted. Although both gasoline and ethanol production may result in negative environmental effects, this study indicates that ethanol production traced through a supply chain may impact less area and result in more easily reversed effects of a shorter duration than gasoline production.

Keywords Biofuel · Transportation · Supply chain · Sustainability · Time · Space

Introduction

Energy sources that can meet the demands of current and future generations without causing unacceptable environmental consequences are vital (Greene and others 2010; NSB 2009). Bioenergy in the form of liquid fuel has the potential to reduce dependence on petroleum while simultaneously reducing carbon dioxide (CO_2) emissions that influence global climate (Robertson and others 2008). Decision makers at local, regional, national, and global levels are seeking to understand resource demands and potential environmental effects from bioenergy production relative to those of traditional, non-renewable sources. We postulate that a holistic, multi-scale comparison of energy production and associated environmental effects provides a way to understand these effects and then implement energy options designed to protect and preserve resources for future generations.

Gasoline and ethanol are likely to be part of the world's transportation fuel options for several decades, in part because the majority of automobiles require energy-dense liquid fuels (Fairly 2011). The United States (US) Energy Independence and Security Act (EISA) of 2007 mandates that the energy-equivalent of 136 billion L of ethanol from renewable sources be blended into transportation fuel by

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2022. The European Union (EU) has agreed to replace 10 % of its transportation fuels with renewable sources by 2020 (EU 2012). At least 47 countries were producing fuel ethanol in 2010. However, the total volume of ethanol produced worldwide (88 billion L in 2010) is still substantially less than that of gasoline (2,281 billion L in 2010) [calculations based on EIA (2011)], and petroleum-based gasoline is expected to remain the primary fuel source for light duty vehicles until at least 2035 (USDOE 2010).

Gasoline production and consumption currently involve many more countries and larger scales of export and distribution than ethanol production and consumption; however, ethanol production is expanding. In 2010, approximately half of the world's countries produced and all countries consumed gasoline (EIA 2011). The top ten gasoline producers—Saudi Arabia, Russia, the US, China, Iran, Canada, Mexico, the United Arab Emirates, Brazil, and Nigeria—are scattered across the globe and produce about 38 % of the world's total gasoline supply by volume (EIA 2011). Approximately 35 % of the global gasoline supply was consumed by three nations: the US, China, and Japan. Nearly 50 countries produced and consumed fuel ethanol in 2010, but the US and Brazil were by far the largest producers of ethanol (89 % by volume) and consumers of ethanol (86 % by volume) [calculations based on EIA (2011)]. Estimates suggest that worldwide production of ethanol may nearly double by 2020 as compared to 2010 (OECD-FAO 2011).

This article examines several scales associated with the production of ethanol and gasoline to compare the potential environmental effects (both positive and negative) that may be the most critical to understanding differences between the two fuel supply chains. For this analysis, we define "scale" as the lower and upper bounds of spatial extent and temporal duration associated with a process step or its related effects. Our underlying hypothesis is that environmental effects of different fuel supply-chain steps may be unique at different spatial extents and that understanding the differences in duration of environmental effects is also critical when comparing fuels. We synthesize the key scale differences of anticipated environmental effects in tabular form and on space–time diagrams to give an overview of the types of environmental tradeoffs that may be involved in shifting from gasoline to ethanol blends. We limit this analysis to ethanol production from biomass feedstocks (i.e., traditional agricultural crops, such as corn and sugar cane, agricultural and forest residues; and dedicated energy crops, such as switchgrass, miscanthus, energy cane, and energy sorghum). We also encourage future research that examines potential environmental effects from other ethanol sources (e.g., municipal solid waste) and from the production of biodiesel (e.g., from soy, algae).

Methods Used for Fuel Supply Chain Comparison

This comparative analysis of ethanol and gasoline production and associated environmental effects is based on an extensive literature review. We are not aware of any previous reviews or summaries of the temporal and spatial dimensions of complete supply pathways for gasoline or of any systematic comparisons of gasoline and ethanol in terms of relative environmental costs and benefits across time and space. We compare the space–time dimensions of the production and environmental effects from gasoline and ethanol production supply chains as they exist today, with current assumptions about near-term technology development and production volumes.

Some data related to environmental effects are not publically available, even for the well-established petroleum industry. Therefore, the examples contained in this article are not necessarily comprehensive, and the estimates provided in this article are not necessarily conclusive. At present, environmental effects associated with expanded production of bioenergy crops are not well understood (Rowe and others 2009), necessitating simulation of commercial-scale bioenergy systems to estimate their potential environmental effects (e.g., Fernando and others 2010). While some information on the scale of direct environmental effects of gasoline production is accessible, little analogous information on indirect effects is available.

We have organized our comparative analysis and discussion according to six major fuel supply-chain steps (Fig. 1): (1) establish fuel sources, (2) obtain raw material, (3) distribute raw material to refineries, (4) convert raw material into fuel, (5) distribute fuel, and (6) use fuel for transportation. In addition to preparing summary tables of the scales of environmental effects found during our literature review, we have synthesized the key scales of fuel production processes and associated potential environmental effects on space–time Stommel diagrams (Vance and Doel 2010). Our Stommel diagrams are intended (1) to increase awareness of important scale differences that need to be considered when comparing the environmental effects of ethanol and gasoline production and (2) to increase awareness of how scale can influence sustainability assessments in general.

Stommel diagrams have a longstanding importance in the discipline of landscape ecology, a field of study partially motivated by the need to understand the characteristic spatial and temporal scales of ecological events related to disturbances (Turner and others 2001). Stommel diagrams have proven useful for synthesizing the main concepts and patterns of an emerging field of study to make predictions and aid in management (Vance and Doel 2010). We use these diagrams to summarize the potential upper and lower

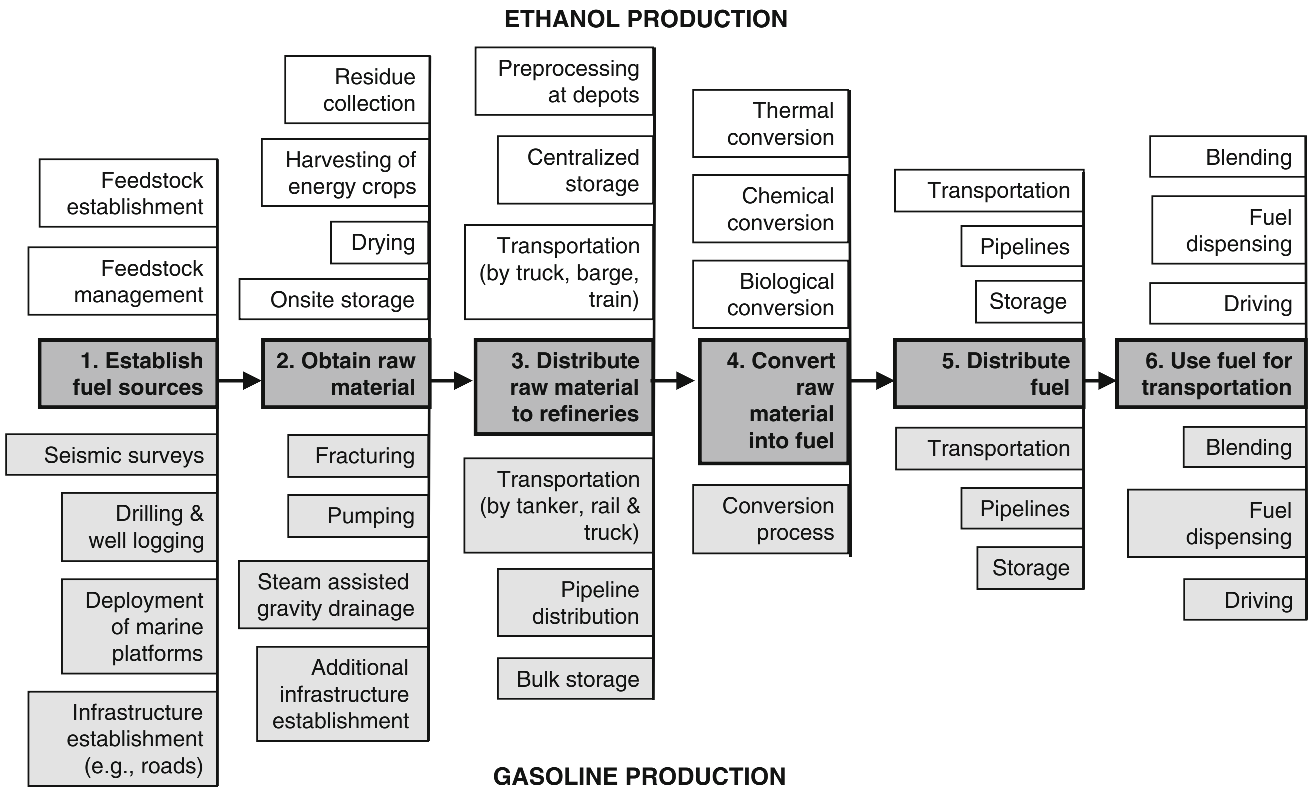


Fig. 1 Comparison of potential gasoline and ethanol-supply-chain steps

bounds of the spatial extent and duration for each process step and the associated environmental effects. Box boundaries on our Stommel diagrams should be considered fuzzy because of the uncertainty surrounding the limited scientific information in the literature. Moreover, when shares of effects are allocated to coproducts (e.g., natural gas produced along with crude oil, chemical products associated with gasoline production, dried distillers grains (DDGs) or electricity associated with bioethanol production), the quantification and scale of effects can vary. In addition to scale, the issues of co-product allocation and causal attribution are among the factors that have complicated efforts to develop comparable life-cycle assessments for fossil and bioenergy fuel production pathways (Wang and others 2011; Kline and others 2011); we do not address these factors in this analysis.

The current body of literature reveals that the spatial extent of supply-chain steps and their environmental effects can be conceptualized in several ways: (1) based on a particular operation (e.g., a feedstock production system, a specific oil well); (2) based on cumulative area of one operation within a region (e.g., the "fuelshed" area supplying a bioenergy refinery, the petroleum-rich Bakken geologic formation in the western US and Canada); (3) based on the cumulative area currently used or affected by a given operation (e.g., all agricultural fields or geologic formations that currently contribute to the global fuel supply); or, (4) based on the total global area that *could* be used or affected by an energy production operation in the future. The spatial extent selected for analysis influences the relevant temporal scale for analysis and vice versa. We highlight examples from all four perspectives and synthesize findings on our Stommel diagrams to demonstrate the potential range of scales associated with the subprocesses and effects pertaining to each fuel-supply-chain step (Fig. 1).

Step-by-Step Scales Comparison of Fuel Production Processes and Environmental Effects

This section discusses the key findings from our literature review and is organized according to the six fuel production steps and sub-steps depicted in Fig. 1. The key environmental effects and their associated spatial extents and temporal durations are summarized for gasoline (Table 1) and ethanol (Table 2) according to the same six steps. The final results of our comparative analysis are presented on Stommel diagrams that synthesize and compare the scales of the fuel production processes (Fig. 2a, b) and their

Table 1 Spatial extent and temporal duration of potential environmental effects associated with gasoline production at each step of the liquid transportation fuel supply chain. Greenhouse gas emissions are treated as a cross-cutting effect

Supply-chain step	Gasoline production subprocess	Potential environmental effects	Extent summary	Duration summary
1. Explore for oil	Seismic surveys	Disturbance of ecologically sensitive areas including wetlands and tundra[1] Increased access to formerly remote areas for hunting (or poaching) and use of off-road vehicles[2] Loss of natural vegetation, either directly or indirectly[3] Introduction of new species (beneficial, pest or invasive)[4] Functional habitat loss,[5] habitat fragmentation,[6] and habitat avoidance[7] Reduced population densities of birds and mammals[8] Alteration of predator–prey relationships[9] Damage to aquatic systems from increased sediment or changes to drainage patterns[10] Soil damage through compaction and/or mixing of soil horizons[11]	Field–region	Days–centuries
	Drilling and well logging	Perforations in cap rock formations[12] Habitat loss and fragmentation[13] Animal avoidance of wells, infrastructure[14] Air and groundwater contamination from disposal of drill cuttings[15] Generation of radioactively contaminated waste streams such as process water, mud and equipment[16] Terrestrial surface water contamination from oil spills and sedimentation[17] Marine-oil spills,[18] including impacts on coastal wetlands[19]	Field–region	Months–millennia
	Deployment of marine platforms	Noise impacts on whales[20] Seabird mortality from collision, oiling, incineration by flare[21] Structural effects on marine life (e.g., designation as artificial reefs,[22] adverse impacts[23])	Field–region	Weeks–centuries
	Infrastructure establishment (e.g., roads)	Habitat fragmentation[24] Invasion by plant species[25] Decline in aquatic macroinvertebrate density and taxonomic diversity due to siltation[26] Hydrologic alteration through longterm surface water mining for ice roads[27] Thawing of Arctic permafrost with associated thermokarst formation and flooding;[28] potential release of carbon stores into the atmosphere;[29] and gravel extraction for prevention[30]	Field–region	Weeks–decades

Table 1 continued

Supply-chain step	Gasoline production subprocess	Potential environmental effects	Extent summary	Duration summary
2. Extract oil	Fracturing	Alteration of groundwater flow and quality[31] Alteration of river flow[32]	Field–region	Decades–centuries
	Pumping	Surface and subsurface contamination from improperly abandoned wells[33] Seismic events[34] Coastal land subsidence[35] Generation of produced water containing toxics and radioactive materials[36] Bird fatalities in produced water ponds[37] Plant and soil toxicity due to brine spills[38] Bird, fish and mammal fatalities due to marine-oil spills[39] Fires from terrestrial oil spills[40] Loss of mangroves and fish habitat due to oil runoff into waterways during rain events[41] Loss of saltmarsh vegetation from oil spills[42] Air pollution from flaring[43] Permanent depletion of subsurface deposits of petroleum	Landscape–globe	Months–millennia
	Additional infrastructure establishment	Habitat fragmentation[44] Loss of wetlands and/or habitat[45] Invasion by non-native plant species[46] Species decline[47] Animal avoidance[48]	Field–landscape	Years–decades
3. Distribute crude oil	Transportation (ocean tanker, rail and/or truck)	Marine-oil spills[49] Aquatic and shoreline biological effects of spills (e.g., wetland vegetation;[50] fish;[51] benthic invertebrates;[52] marine mammals[53])	Field–globe	Hours–decades
	Pipeline distribution	Land clearing[54] Disturbance of remote areas such as the North Slope tundra[55] Biological effects of spills[56]	Field	Hours–decades
4. Produce gasoline	Conversion process	Air pollution[57] Water pollution[58] Soil pollution[59] Radioactive solid waste streams due to buildup of naturally occurring radioactive materials[60]	Field–neighborhood	Hours–centuries

Table 1 continued

Supply-chain step	Gasoline production subprocess	Potential environmental effects	Extent summary	Duration summary
5. Distribute gasoline	Transportation (truck, rail)	Air pollution[61] Gasoline spills	Field–region	Minutes–years
	Pipelines	Freshwater spills from ruptures leading to fish kills and species fragmentation[62] Toxicity of spills to terrestrial plants and soils[63]	Field–landscape	Months–decades
	Storage	Toxicity of spilled gasoline to aquatic and terrestrial plants and animals[64] Leaking of USTs and associated groundwater contamination[65] Evaporative emissions	Field	Hours–decades
6. Use gasoline	Fuel blending	Gasoline spills Evaporative emissions	Field	Minutes–hours
	Fuel dispensing	Evaporative emissions Leaking of USTs and associated groundwater contamination[66]	Field	Hours–decades
	Driving	Gaseous and particulate emissions[67]	Neighborhood–continent	Minutes–weeks
All Steps	Greenhouse gas emissions	Warming atmosphere and associated changes in Earth's climate[68]	Globe	Decades–centuries

[1] Schneider (2002) and Thompson (2011); [2] Schneider (2002); [3] Bayne and others (2005a), Drawe and Ortega (1996), Lee and Boutin (2006), MacFarlane (2003), Rabanal and others (2010), Rich and others (1994), and Schneider (2002); [4] MacFarlane (2003) and Schneider (2002); [5] Dyer and others (2001); [6] Lee and Boutin (2006), Bayne and others (2005a), and Schneider (2002); ; [7] Rabanal and others (2010) and Schneider (2002); [8] Benítez-López and others (2010); [9] Schneider (2002); [10] Schneider (2002); [11] Schneider (2002); [12] Nordbotten and others (2009) and Miskimins (2009); [13] Cronin and others (1998), Finer and Orta-Martinez (2010), Schneider (2002), and van Dyke and Klein (1996); [14] Dyer and others (2001); [15] Hall and Spell (1991) and Zimmerman and Robert (1991); [16] USEPA (2010a) and IAEA (2003); [17] Hyland and others (1994) and UNEP (2011); [18] County of Santa Barbara (2005) and National Commission (2011); [19] Mendelssohn and others (2012); [20] Richardson and others (1990); [21] Wiese and others (2001); [22] National Commission (2011) and Gates (2012); [23] Iversen and Esler (2010); [24] CONAP (2006); [25] Simmers and Galatowitsch (2010); [26] Couceiro and others (2010); [27] Pelley (2001); [28] Walker and others (1987); [29] Trucco and others (2012) and Turetsky and others (2002); [30] Jorgenson and Joyce (1994) and Walker and others (1987); [31] Manual (2010) and Osborn and others (2011); [32] Vaht and others (2011); [33] Kelm and Faul (1999), Miskimins (2009), and IOGCC (2009); [34] NRC (2012); [35] Morton and others (2006), Chilingar and Endres (2005), Hettema and others (2002), and Nagel (2001); [36] Veil and others (2004) and Khatib and Verbeek (2003); [37] Ramirez (2010); [38] API (1997), Bass (1999), Efroymson and others (2004a), and Jager and others (2005); [39] Hussain and Gondal (2008) and Khordagui and Al-Ajmi (1993); [40] UNEP (2011); [41] UNEP (2011); [42] Lin and Mendelssohn (2012); [43] Schneider (2002); [44] CONAP (2006); [45] Ouyang and others (2008); [46] Simmers and Galatowitsch (2010); [47] Couceiro and others (2010) and Holloran and others (2010); [48] Haskell and others (2006) and Lyon and Anderson (2003); [49] Lucas and MacGregor (2006) and Redondo and Platonov (2009); [50] Mendelssohn and others (2012); [51] Neff and others (1985); [52] Neff and others (1985); [53] DeGange and others (1994); [54] Couceiro and others (2010); [55] National Commission (2011); [56] e.g., zooplankton, Fefilova (2011) and Mendelssohn and others (1990); [57] Gariazzo and others (2005) and Sorkin (1975); [58] Oviatt and others (1982); [59] Maila and Cloete (2004); [60] Gray (1990); [61] USEPA (2010b); [62] Niemi and others (1990) and Kubach and others (2011); [63] Wang and others 1998 and Michel and others (2009); [64] Li and McAteer (2000); [65] USEPA (2012); [66] USEPA (2012); [67] Balat (2011) and Greene (2010); [68] IPCC (2007)

Table 2 Spatial extent and temporal duration of potential environmental effects associated with ethanol production at each step of the liquid transportation fuel supply chain. Greenhouse gas emissions are treated as a cross-cutting effect

Supply-chain step	Ethanol production subprocess	Potential environmental effects	Extent summary	Duration summary
1. Establish biomass feedstock	Feedstock establishment	Loss of natural vegetation, directly or indirectly[1] Change in habitat suitability and species richness[2] Introduction of new species (beneficial, pest or invasive)[3] Changes in soil quality[4] Changes in carbon sequestration[5]	Field–landscape	Months–decades
	Feedstock management (e.g., cultivation, chemical application, irrigation)	Alteration in land management that affect fire regime, nutrient cycles and emissions[6] Air pollution[7] Changes in eutrophication and hypoxia[8] Water quality[9] Groundwater depletion through irrigation[10]	Field–region	Hours–centuries
2. Harvest and collect biomass	Residue collection	Soil erosion[11] Nutrient losses and use efficiency[12] Changes in water quality[13] Changes in forest fire cycle and effects due to fuel management and forest thinning[14]	Field–landscape	Years–centuries
	Harvesting of energy crops	Soil erosion[15] Changes in species richness[16]	Field–landscape	Years–centuries
	Onsite storage and drying of biomass	Gaseous emissions from decomposing biomass[17]	Field	Weeks–years
3. Distribute biomass	Transportation (truck, barge, train)	Gaseous emissions from trucks and barges[18]	Field–region	Minutes–decades
	Preprocessing at depots	Gaseous emissions from machinery[19]	Field	Hours–years
	Centralized storage	Gaseous emissions from decomposing biomass[20]	Field	Weeks–months
4. Produce ethanol	Thermal conversion Chemical conversion Biological conversion	Air pollution, including criteria pollutants regulated under the US Clean Air Act[21] Water pollution[22] Groundwater competition[23] Solid waste generation[24]	Field–neighborhood	Hours–decades
5. Distribute ethanol	Transportation by truck and rail	Air pollution resulting from gaseous and particulate emissions[25] Ethanol spills[26]	Field–region	Minutes–years
	Pipeline	Ethanol spills[27] Soldering waste	Field–region	Hours–months
	Storage	Ethanol spills[28] Evaporative emissions	Field	Hours–months

Table 2 continued

Supply-chain step	Ethanol production subprocess	Potential environmental effects	Extent summary	Duration summary
6. Use ethanol	Blending and dispensing	Ethanol and/or gasoline spills[29] Evaporative emissions Leakage from storage tanks[30]	Field	Seconds–decades
	Driving	Gaseous and particulate emissions[31] Leakage from vehicle tanks and hoses	Neighborhood–continent	Minutes–weeks
All Steps	Greenhouse gas emissions	Warming atmosphere and associated changes in Earth's climate[32]	Globe	Decades–centuries

[1] Fargione and others (2008); [2] Robertson and others (2011), Sage and others (2010), and Dhondt and others (2007); [3] Quinn and others (2010); [4] Nijsen and others (2012); [5] Tolbert and others (2002); [6] Kline and Dale (2008); [7] Rettenmaier and others (2010); [8] Dale and others (2010b), Rabalais and others (2002), and Rettenmaier and others (2010); [9] Parish and others (2012); [10] Wu and others (2009); [11] Thomas and others (2011) and Huggins and others (2011); [12] Thomas and others (2011) and Huggins and others (2011); [13] Thomas and others (2011); [14] Kocoloski and others (2011); [15] Nelson and others (2004) and Huggins and others (2011); [16] Roth and others (2005); [17] Emery and Mosier (2012); [18] USEPA (2010b); [19] USEPA (2010b); [20] USEPA (2010b); [21] Archer (2005), Wang and others (2007), and Hess and others (2009b); [22] USEPA (2010b), Evans and Cohen (2009), Levin and others (2002), and Pate and others (2007); [23] Scown and others (2011) and Wu and others (2009); [24] USEPA (2010b); [25] USEPA (2010b); [26] USEPA (2009a, b, 2010b), Powers and others (2001), and Ruiz-Aguilar and others (2002); [27] USEPA (2009a, b), Powers and others (2001), and Ruiz-Aguilar and others (2002); [28] USEPA (2009a, b), Powers and others (2001), and Ruiz-Aguilar and others (2002); [29] USEPA (2009a, b), Powers and others (2001), and Ruiz-Aguilar and others (2002); [30] USEPA (2009a, b), Powers and others (2001), and Ruiz-Aguilar and others (2002); [31] Ginnebaugh and others (2010), Graham and others (2008), Yanowitz and McCormick (2009), and Niven (2005); [32] IPCC (2007)

potential environmental effects (Fig. 2c, d). Because the environmental effects from fuel distribution and end use (Steps 5 and 6) have generally similar spatial extents and temporal durations for ethanol and gasoline (as summarized in Tables 1, 2), we have removed Steps 5 and 6 from the Stommel diagrams; this facilitates a clearer view of the steps that have more pronounced differences in scales.

Both fuel supply chains have the potential to generate greenhouse gas (GHG) emissions at each step of production. The Intergovernmental Panel on Climate Change (IPCC 2007) has concluded that anthropogenic emissions of CO_2 are causing Earth's atmosphere to warm to the extent that global changes in climate are very likely to occur for more than a century. Because the spatial extent and long duration of these global climate change impacts dwarfs the extent and duration of many other potential environmental effects from fuel production, we have deliberately excluded GHG effects from the Stommel diagrams (Fig. 2) and have considered them as a cross-cutting process depicted at the end of each summary table (Tables 1, 2).

Step 1: Establish Fuel Sources

Establishing liquid transportation fuel sources involves locating petroleum reserves from which gasoline may be derived, and selecting and managing feedstocks for ethanol production. Gasoline and ethanol have different environmental effects largely because of the extreme differences in the time cycle and spatial extent associated with the two fuel-source establishment processes (Fig. 2a, b).

Scales of Fuel-Source Establishment Processes

Gasoline and ethanol are both derived from organic matter. Gasoline is derived from crude petroleum reservoirs that pooled in sedimentary rocks beneath Earth's oceans and continents over millions of years as organic material was heated and pressurized under a unique set of climatic and geological conditions. Consequently, gasoline feedstocks were formed many years before human extraction and cannot be replenished on a human timescale. In contrast, biomass grown or residues collected for ethanol production may be replanted or regenerated within a timeframe of months to years across arable lands in the form of crops, trees, crop residues or wood wastes. Under reasonable assumptions for land management and climate change, biomass regrowth and collection cycles can continue indefinitely even if the availability or demand for particular feedstocks changes over time. Thus, the timescales associated with the two feedstock establishment cycles are profoundly different.

The spatial extents and locations of petroleum reserves and ethanol feedstocks also differ substantially. Although

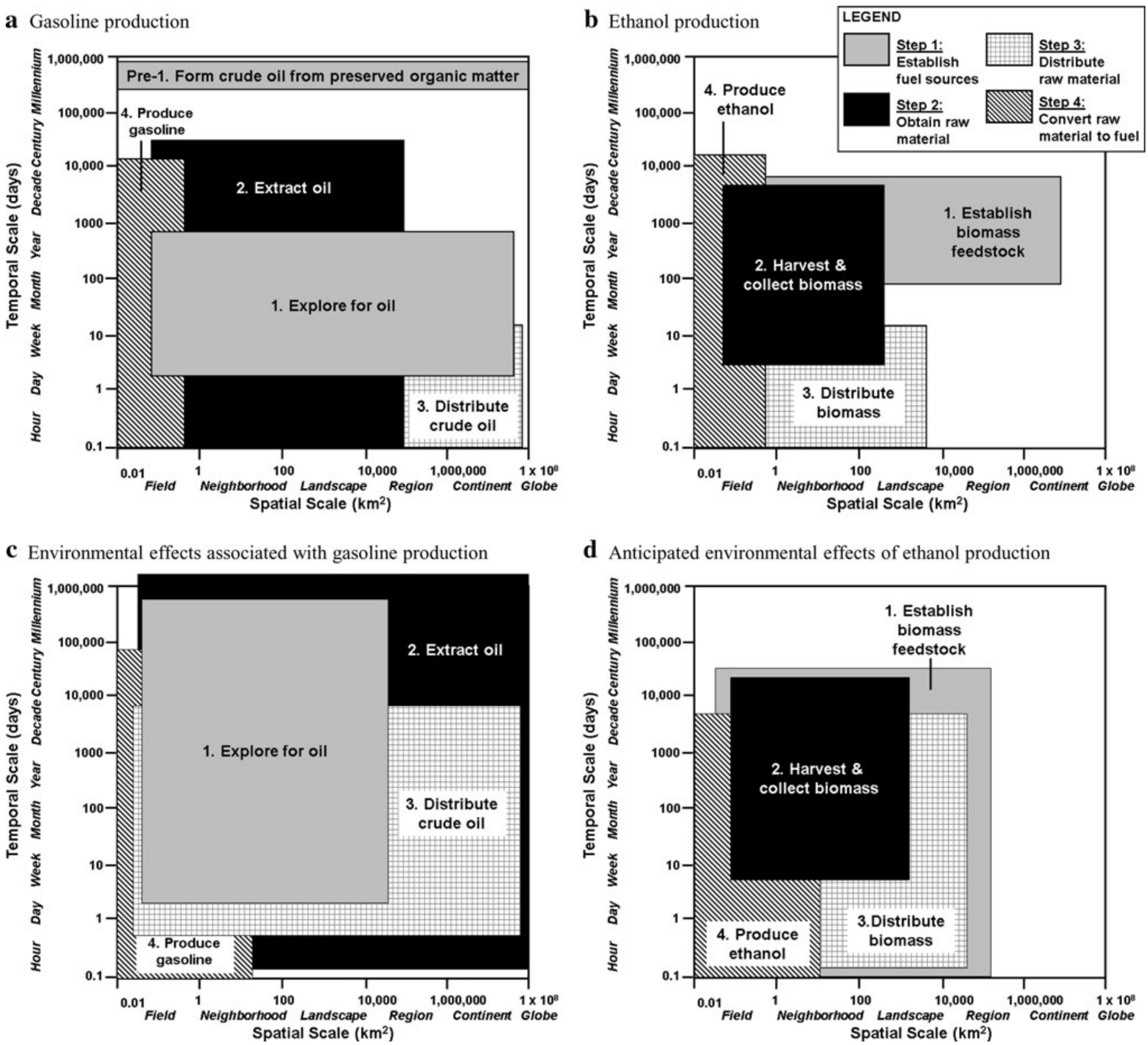


Fig. 2 Stommel diagrams comparing the combined spatial extent and duration of gasoline production (**a**) and ethanol production (**b**), the environmental effects associated with gasoline production (**c**), and the anticipated environmental effects of ethanol production (**d**)

gasoline production volumes are modulated by market demand, geologic factors generally determine the lands, waters, and scales of petroleum exploration. Major oil reserves are still being discovered, and exploration is growing rapidly in some of the most remote and fragile ecosystems on Earth, including the boreal forests of Russia and Canada, the tropical forests and savannas of central Africa, the wetlands and seas of Myanmar and Southeast Asia, and the Peruvian Amazon (Orta-Martinez and Finer 2010, Thompson 2011). With emerging technology, virtually the entire globe—from the Arctic to deposits deep below the oceans—is open to petroleum exploration.

The extent and location of biomass-production systems for ethanol, by contrast, are inherently limited to arable land areas (i.e., in temperate or tropical climates with suitable soils) and are determined through a combination of biophysical, economic, political, and social factors, with individual farmers often making crop decisions and markets determining whether crops are used for biofuels. Although there are concerns that biofuels expansion may compete with the existing food and fiber industries, most dedicated bioenergy production is likely to occur on the 500–5,000 million hectares (Mha) of previously cleared and underutilized land that is already available at a relatively low cost (Kline and others 2009; FAO IIASA 2007). Assessments of potential biomass production consistently point to Africa and Latin America as two regions with a great capacity for expanding dedicated bioenergy feedstock production using resources that do not compete with food production (Lynd and Woods 2011).

Scales of Environmental Effects of Oil Exploration

Gasoline and ethanol production have distinct environmental effects that translate to the different extents and durations summarized in Tables 1 and 2 and Fig. 2c, d. Key effects of oil exploration include landscape fragmentation and the generation of toxic, hazardous, and potentially radioactive waste streams. Although petroleum exploration and agricultural production each have the potential to fragment the surface landscape, the petroleum exploration activities of deep drilling and seismic testing have effects that extend beyond the bounds of arable land (e.g., to oceans and frozen tundra) and include a subsurface dimension.

The spatial extent of seismic testing for oil and gas (O&G) ranges from a few shot holes, (depressions in which explosives are set) in a field to vibrator truck tracks cut across an entire region. Seismic surveys cut trails through natural vegetation, including forests, grasslands, tundra, and other potentially ecologically sensitive areas, making seismic exploration the major driver of landscape fragmentation by the petroleum sector (Lee and Boutin 2006; Bayne and others 2005a; Schneider 2002). Seismic lines are typically straight paths of cleared vegetation ranging in width from 1.2 m to over 12 m (Bayne and others 2005a; Lee and Boutin 2006; MacFarlane 2003; Rabanal and others 2010; Schneider 2002). Spacing between seismic lines generally ranges from under 50 m to 5 km, and tight spacing of 50–80 m may cover areas greater than 100 km^2 when steam-assisted gravity drainage is used to enhance oil recovery (Bayne and others 2005a; Tankard and others 1995).

The cumulative footprint of seismic lines and associated roads and exploratory wells can be extensive. For instance, Finer and Orta-Martinez (2010) calculated that nearly half of the Peruvian Amazon has been physically disturbed by O&G concessions, including more than 104,000 km of seismic lines and 679 wells. Finer and Orta-Martinez (2010) anticipate that a second peak in oil exploration may generate 20,000 km of new seismic lines and 180 new exploratory wells in remote forests of the Peruvian Amazon. In Canada's "Green Zone," an area that comprises 53 % of Alberta's total land area and primarily consists of provincially-owned forest land, the number of trees cut for seismic operations was roughly equal to the number harvested by the forest industry from 1997 to 2001 (Schneider 2002).

Regrowth of vegetation following seismic operations proceeds at rates that depend on latitude, precipitation, nutrient availability, soils, and characteristics of initial disturbance and subsequent use. This vegetation can regrow within 2–3 years in tropical areas with excellent soils and rainfall (Drawe and Ortega 1996). However, recovery is much slower at higher latitudes or following severe disturbances. For example, in the aspen, white spruce, and lowland black spruce forests in Canada's western Boreal Plains, only 8.2 % of seismic lines had recovered more than 50 % of their woody vegetation cover after 35 years (Lee and Boutin 2006). Incomplete regeneration of forests following seismic activities may result in a progressive loss of mature forest and alteration of forest structure (Schneider 2002). Land cleared for seismic surveys is frequently converted to more permanent vehicular tracks and roads. For instance, in Canada's western Boreal Plains, about 20 % of seismic lines became vehicular tracks (some used for off-road vehicles and hunting), and 5 % transitioned to other anthropogenic features such as roads, pipelines, buildings, and timber-harvest blocks (Lee and Boutin 2006; Schneider 2002).

Seismic lines cut through natural ecosystems can generate environmental impacts that extend beyond the direct footprint because of "edge effects" (MacFarlane 2003) and functional habitat loss (Dyer and others 2001). Seismic lines are associated with invasive plant species (MacFarlane 2003) and loss of functional habitat for elk (Dyer and others 2001), and effects on flora and fauna may persist for many years (Rich and others 1994). A meta-analysis of 49 wildlife studies (Benítez-López and others 2010) found that the effects of linear features such as seismic lines on population densities generally extend out to 1 km for birds and 5 km for mammals. Warbler populations in the boreal forest of western Canada were found to decrease as seismic-line density increased above 8.5 km/km^2 (Bayne and others 2005b). Schneider (2002) also documented wildlife disturbances in Canada due to land clearing and dynamite explosions associated with seismic testing, particularly during periods of caribou calving, nesting or low food supplies.

The installation of exploratory oil wells can lead to environmental impacts such as habitat fragmentation and animal avoidance (van Dyke and Klein 1996; Dyer and others 2001). In Alberta, Canada, close to 0.075 km^2 of forest disturbance occurred for each well drilled (Schneider 2002), and caribou avoided areas at a distance of up to 1 km from well platforms (equivalent to an area >3 km^2) (Dyer and others 2001). However, in a study in the Prudhoe Bay Oil Field in northern Alaska, caribou did not avoid oilfield infrastructure (Cronin and others 1998).

Ice roads and well pads constructed to access the North Slope oilfields during the winter exploration season in Alaska are likely to have a cumulative impact on the hydrologic cycle of Alaska's coastal plain (Pelley 2001; Angles 2011). Large volumes of water on the order of 1.3–2.5 million L/km are used to create this temporary infrastructure (Cott and others 2008). Typically the water is sprayed over an aggregate of ice chips obtained from the surrounding area. Permits allow oil companies to remove 15 % of the liquid volumes available in the surrounding tundra lakes for infrastructure and drilling operations (NSDSS 2012); these snowfed lakes can take two years to refill (Pelley 2001). As the weather warms, the ice infrastructure tends to melt into other watersheds rather than being returned to the source watershed (Pelley 2001).

A reliable and comprehensive accounting of all wells drilled in the quest for fossil fuels could not be identified. Reports of recent drilling and average oil well densities exist but may be misleading as they often omit abandoned wells and "dry holes." Well-density statistics vary with the total area being considered. North American well densities have been reported to range from 0.3 to 2.4 wells per km^2 (Nicot 2009; Gasda and others 2004; BLM 2004), and wells are drilled across the globe at densities of up to 6 wells per km^2 (IPCC 2005). Well-density data need to be considered in concert with land area impacted to assess spatial and temporal effects. By one of the US estimate, each well disturbs 3.64 ha (9 acres) of land, including land used for roads (BLM 2012).

The cumulative area devoted to well pads across a landscape may be minimized via directional drilling of multiple wells from the same pad. For instance, the typical single horizontal well pad drilled in the Bakken formation of North Dakota has an area of 0.016–0.024 km^2, but several horizontal wells may be drilled from a well pad that is only slightly larger (i.e., with an area of 0.020–0.028 km^2) (North Dakota 2012). Once the wells are producing, it may be possible to reduce the total well pad area [e.g., by approximately 25 % in the Bakken formation (North Dakota 2012)]. However, the cumulative area of other infrastructure (e.g., roads, pipelines) typically increases once the wells are producing.

While O&G wells are being installed, special fluids referred to as "mud" are used to facilitate the drilling process. Drilling fluids often contain additives such as chromium, barium or chlorides (Hall and Spell 1991) that help counteract and control pressures encountered in the borehole. A typical 3,000 m-deep well requires an input of 300–600 t of mud and produces 1,000–1,500 t of drill cuttings, or waste material containing mud mixed with rock bits and hydrocarbons (E&P Forum and UNEP IE 1997). The size of the bore hole impacts the amount of solid waste produced; for instance, a 31-cm (12.25-in.) borehole produces 96 % more waste than a 22-cm (8.75-in.) borehole (Hall and Spell 1991). Drill cuttings may be incinerated, landfilled, or landfarmed on site (Zimmerman and Robert 1991).

Landfarming is a bioremediation treatment that involves diluting the drill cuttings with in situ soils and periodically tilling the soil to aerate it and to promote microbial degradation of hydrocarbons and chemical additives. This process is typically the cheapest and most widely used method of disposal for drill cuttings since it allows native soil micro-organisms to degrade the hydrocarbons and natural leaching action to reduce the chlorides (Zimmerman and Robert 1991). However, runoff from treatment areas during rain events can generate a large volume of wastewater that may cause harm in the absence of other forms of treatment (Hall and Spell 1991). The extent over which drill cutting waste is spread on land and the composition of the underlying soil partly determine the environmental effects of drilling. In 33 landfarm sites throughout Alberta, Canada, the average ratio of cuttings-to-surface area was 45,342 m^3/km^2, and reclamation target conditions were met in an average of 2–4 years (Zimmerman and Robert 1991).

Petroleum exploration has the potential to generate long-term radioactive wastes that may adversely affect human health or wildlife. Drilling through rock formations containing naturally occurring radionuclides such as uranium and thorium can bring to the surface decay products such as radium-226, radium-228, and radon-222 in the form of process water or gas and thereby contaminate equipment, evaporation ponds, pits, and other storage areas with radium-contaminated water, drilling mud, sludge, and slimes (USEPA 2010a; Veil and others 2004; Tan and Pelletier 2009; Gray 1990). Radon gas has a short half-life (3.8 days) but decays to lead-210 with a half-life of 22 years. Radium-226 has a half-life of 1,600 years and is often in process water (Tan and Pelletier 2009). An additional source of radioactive waste from the O&G industry is generated through the manufacture, storage, transportation, and disposal of ionizing sources (e.g., cesium-137, americium-241) used in exploratory well logging tools and associated calibration equipment (IAEA 2003). Radioactive waste streams may require special management—sometimes over long time periods—in order to prevent increased risk of cancer in humans and other organisms.

Marine extraction and crude oil spills that occur during petroleum exploration are problematic for birds and mammals. Routine offshore operations can affect whales and seabirds over time (Wiese and others 2001; Richardson and others 1990), and large marine-oil spills may acutely impact thousands of vertebrates (Loughlin 1994). Two of the largest marine spills in US history occurred because of well blowouts during ocean drilling (Table 3). Marine well blowouts create underwater plumes of oil droplets and surface slicks over large areas and eventually contaminate shorelines as winds and currents move the oil. The risk of accidental releases of hydrocarbons associated with deep-water drilling typically increases with distance from the shoreline and increased depth (National Commission 2011). Offshore and deepwater drilling for petroleum are expected to become more prevalent in future years as

Table 3 The three largest documented marine-oil spills in the United States history

Date	Name	Location	Cause	Volume released (millions of L)	Water area impacted (km^2)	Coastline length impacted (km)	Duration of impacts
Jan. 1969	Santa Barbara Channel	California coast	Well blowout	13–16[1]	2,072[1]	129[1]	Several years[1]
Mar. 1989	Exxon Valdez	Prince William Sound, Alaska	Grounded tanker	41[2]	28,490[3]	2,092[3]	Decades[4]
Apr. 2010	Deepwater Horizon	Gulf of Mexico	Well blowout	779[5]	88,522[6]	1,046[7]	Unknown[8]

[1] County of Santa Barbara (2005) and National Commission (2011); [2] There is uncertainty surrounding volumes involved in blow-outs and spills. The reported amount of oil spilled by *Exxon Valdez*, approximately 41 million L (Cleveland 2010b), was disputed based on later research suggesting that the actual amount of oil spilled was between 113 and 132 million L, about triple the commonly cited amount (NPR 2010); [3] Cleveland (2010b); [4] Oil slicks were still present more than 20 years later (Biello 2010). Hydrocarbons remained in sediments as late as 2007 and are expected to persist for decades (Short and others 2007; Li and Boufadel 2010). From 2001 to 2005, the area of oiled sand was declining at a rate of less than 4 % per year (Short and others 2007). In 2004, several species of waterfowl and fish had not begun to recover (Cleveland 2010b). According to work by Matkin and others (2008) the Exxon Valdez oil spill caused losses of 33 and 41 %, respectively, to two groups of killer whales that had not yet returned to pre-spill numbers 16 years after the event; [5] Mendelssohn and others (2012); [6] This is the total water area that had fishing restrictions following the spill (National Commission 2011); [7] The well depth and distant location from the shore, combined with dispersants, winds and currents kept much of the oil away from the coastline (Cleveland 2010a; National Commission 2011). Mendelssohn and others (2012) reported that 283 km of marsh shoreline was moderately to heavily oiled by this event. The coastal wetlands of the Mississippi River Delta ecosystem are linked to 30 % of the US commercial fishery production and the protection of an oil and gas (O&G) infrastructure that supplies ~1/3 of the US O&G supply and 50 % of the nation's refining capacity (Mendelssohn and others 2012); [8] Recovery of flora and fauna following the Deepwater Horizon oil spill is likely to be quite variable. For instance, a study of two plant species in a Louisiana coastal marsh conducted 7 months after the spill showed that one species (*Spartina alterniflora*) had recovered almost completely while the other species (*Juncus roemerianus*) had not (Lin and Mendelssohn 2012)

petroleum prices increase (Leiby and Rubin, in press), particularly in areas of the Atlantic Ocean rimmed by the Gulf of Mexico, Brazil, and western Africa (National Commission 2011). Thus, marine-oil spills have the potential to occur more frequently in the absence of a concomitant increase of preventative measures.

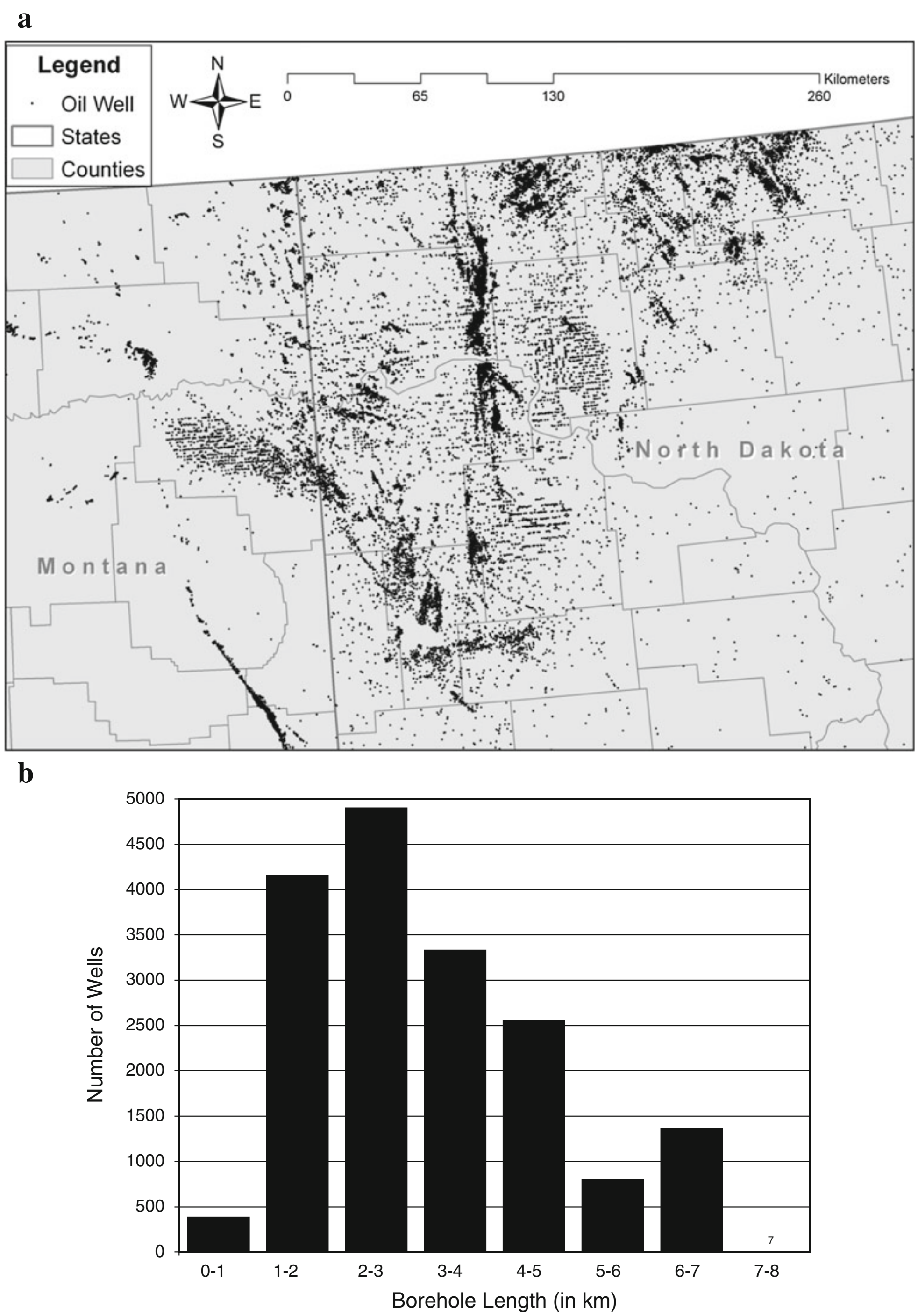
Subterranean effects of O&G exploration are not captured by surface area measures, and their long-term impacts are not completely understood. Drilling occurs to depths of thousands of meters below Earth's surface (for example, see Fig. 3) and alters subsurface pressures through removal of fluids and fracturing of rock formations to increase permeability. The boreholes recorded in the oil field illustrated in Fig. 3 have a cumulative length of over 55,000 km, equivalent to ten times the distance from New York to London [calculations based on North Dakota (2011) and Montana's Board of Oil and Gas (2011)]. "Dry holes," exploratory wells that did not produce crude, represent additional underground disturbances, but there are insufficient data on their location, extent, and effects. Perforations of Earth's crust create linkages among the surface environment, targeted fossil deposits, subterranean aquifers, and other geologic formations, and these new connections may persist for eons (Miskimins 2009). Ubiquitous perforation of rock formations could limit the potential for effectively capturing and storing CO_2 in underground reservoirs, a process that has been proposed to help mitigate global climate change (Nordbotten and others 2009; Stephens 2006).

Scales of Biomass Establishment Processes

At present, most ethanol production is based on corn grain (US) and sugarcane (Brazil) feedstocks that are grown for multiple uses. However, a wide variety of other biomass resources are available for ethanol production depending on regional climate, soils, and existing conditions (USDOE 2011; Dale and others 2011b). Potential feedstocks include grains, perennial grasses, woody crops, crop residues, and forest thinnings. To date, most dedicated energy crop production has occurred on a small scale and at experimental sites; few landscapes integrating large-scale production of dedicated bioenergy crops exist beyond the forest sector.

Worldwide estimates of the amount, type, and location of land available for additional biofuels production vary greatly according to assumptions about crop types, management, yield, climate change, and current and competing land uses (USDOE 2011; HEC and CABI 2010). Accurate estimates of the amount of land available for biomass production are limited by land-cover datasets, which are often derived from satellite imagery. It is difficult to detect land-use allocations and trends, for imagery only documents land cover during recent decades, and alterations in sensors and classification systems can compromise change analysis. Furthermore, the direct effects of establishing new feedstocks (e.g., dedicated cellulosic crops) must be modeled using many assumptions, about which there is limited agreement (CBES 2009). It is even more challenging to

Fig. 3 Oil well locations with borehole length data (a) and the corresponding frequency distribution of subsurface borehole lengths (b) for the portions of northeastern Montana and northwestern North Dakota shown in this *map*. Much of this area is underlain by the Bakken Formation. The 17,540 oil wells shown in this figure have an average borehole length of 3,160 m and are mostly directional or horizontal (rather than vertical). [Based on North Dakota (2011) and Montana's Board of Oil and Gas (2011)]



estimate the extent of indirect land-use change from biofuels expansion. While indirect effects may have a major influence on the perceived environmental effects of ethanol production (Mullins and others 2010; Kline and others 2009), little consensus exists on how to quantify the indirect effects or even on how to determine whether such effects might be positive or negative (Kline and others 2011).

The extent of feedstock production is influenced by the opportunities perceived by growers. Land-use options that consistently offer higher net financial return and lower risk are likely to displace other land uses (Hayes and others 2009). However, even though use of US corn for ethanol production increased five-fold from 2001 to 2009, improvements in corn yields were largely responsible for the overall increase in domestic production of corn, and the domestic market adjusted flexibly to ethanol production with minimal land-use change (Oladosu and others 2011). A study of Brazilian ethanol production from sugarcane

(Goldemberg and Guardabassi 2010) found that combined advances in farm practices and conversion technologies have enabled Brazil to increase ethanol production volumes from approximately 200,000 to 700,000 L per km^2, an increase of 350 % since 1975.

Complications can arise when attempting to attribute a specific area of land to ethanol production (e.g., Oladosu and others 2011). Corn and sugarcane are industrial commodities involving a diversity of producers, production systems, and co-products. The vast majority of US corn is used for animal feed while most Brazilian sugarcane is processed for sugar and molasses. A corn producer does not necessarily know or care whether his output is used for ethanol; the grain is simply sold on the open market. Land area is therefore calculated based on aggregate numbers associated with the amount of ethanol produced. In 2008, the total area of US land used to grow corn for ethanol production was estimated to be 81,300 km^2 or about 5 % of total US cropland, and the area of Brazilian land used to grow sugarcane for ethanol production was 34,000 km^2 or about 5 % of total cropland in Brazil (FAOSTAT 2012; Goldemberg and Guardabassi 2010). However, when co-products such as the DDGs—a high-protein feed ingredient coproduced with corn ethanol—are taken into account, the cropland allocation to ethanol could be 33 % less than what is commonly reported based simply on the total amount of feedstock processed by a corn-ethanol mill (Wallington and others 2012; Kline and others 2011).

Scales of Environmental Effects of Biomass Feedstock Establishment

Environmental benefits or negative impacts may result from land-cover changes and management practices associated with feedstock production depending on the choice of feedstock, extent of production, previous land use, and measures of effects (e.g., Hammerschlag 2006; Robertson and others 2008). Environmental effects from feedstock production can include changes in overall GHG emissions, with global implications for atmospheric warming and associated changes in climate (IPCC 2007), and changes in water quantity, water quality, soil quality, air quality, and biodiversity at the local to regional scale (McBride and others 2011). For example, if natural areas with high carbon stocks such as old-growth forests are changed to annual cropland, then net GHG emissions are likely to increase and biodiversity to be compromised (Fargione and others 2008). In contrast, growing biofuel feedstock could enhance net carbon sequestration and biodiversity on degraded land, and biofuel markets could increase the value of biomass, thereby providing incentives to improve management of forests and land that was previously cleared, abandoned, or burned (Kline and Dale 2008). Quantitative empirical estimates of these environmental effects—particularly changes to biodiversity—are sparse (Ridley and others 2012). In contrast to O&G exploration, no radioactive waste streams are generated by ethanol production; crude oil spills associated with marine extraction also have no analog in ethanol production.

Perennial bioenergy crops have the potential to stabilize soils and replenish soil nutrients within their root systems (Tolbert and others 2002) and, therefore, to regenerate large areas of degraded soils in Africa, Asia, and other regions (Nijsen and others 2012; Lynd and Woods 2011; Lal 2004). If management-intensive annual row crops such as cotton and corn are displaced by perennial herbaceous and woody bioenergy crops, watersheds may experience improved water quality through decreased stream sediment loads and nutrient concentrations (Parish and others 2012). Conversely, if forests, perennial pastures and grasslands are displaced by annual row crops such as corn, increased erosion and reduced water quality could extend beyond the field boundaries (Simpson and others 2008). An EU environmental impact assessment of 15 potential energy crops and their aggregated environmental effects on six categories of indicators (e.g., soil, biodiversity) found that growing dedicated annual energy crops (e.g., rapeseed, sugar beets) does not inflict higher impact on the environment than traditional potato and wheat farming, and that woody and lignocellulosic energy crops have reduced impacts on soil erosion and biodiversity compared to annual cropping systems (Fernando and others 2010).

The extent of water quantity and/or quality effects associated with ethanol production varies depending on the feedstock selected, prevailing site conditions, management techniques, the amount of land under feedstock production, and other pressures in the system. For instance, a case study of the potential effects of *Jatropha* plantations in India showed reduced water availability at the sub-basin scale but improved groundwater recharge at a larger watershed scale (Yeh and others 2011). A very large-scale example is the hypoxic zone along the Louisiana–Texas coast that has ranged in size from 4,400 to 20,000 km^2 as measured during summers from 2000 through 2010 (Rabalais and others 2002; LUMCON 2012). Agricultural practices in the US Corn Belt lead to nutrient runoff into the Mississippi and Atchafalaya rivers that, in combination with seasonal stratification of Gulf waters, precipitation regimes, and other factors, can contribute to hypoxia (low oxygen levels) in the Gulf of Mexico (Dale and others 2010b). Depending on the selected crop type and its management, biofuel cultivation may contribute to or mitigate conditions associated with this regional hypoxia and aquatic eutrophication (Dale and others 2010b).

The extent of effects of feedstock production on biodiversity are a result of field's size as well as its shape, location, prior use, and timing of management actions (including tillage, rotation, harvest, and duration of crop in a field) (Dale and others 2010a). In one biodiversity study, bird species richness was associated with patch size for switchgrass and native prairie but not with corn or landscapes with high forest cover (Robertson and others 2011). An investigation of birds in short-rotation woody crop plantations in New York state found that avian species richness matched that of early successional habitat and tended to increase with age of coppice (Dhondt and others 2007). The researchers recommend large-scale plantings with staggered timing of coppicing to avert potential negative effects on bird diversity (Dhondt and others 2007). There have been relatively few studies of the extent of effects of biofuels production on biodiversity, and this is an area that merits further research (Ridley and others 2012).

Step 2: Obtain Raw Material

The second step in the fuel life cycle involves obtaining the raw material (Fig. 1). For gasoline, this step occurs after a petroleum-bearing rock formation is discovered and deemed economically viable, and when crude oil is extracted from the reservoir—often via multiple wells across a region. For ethanol production, this step takes place when the biomass feedstock is harvested or collected. If petroleum prices remain high (over $100 per barrel), more gasoline is expected to be produced from unconventional oil sources including oil sands (also known as. tar sands), oil shales that require surface mining, water-intensive hydraulic fracturing, and ultra-deepwater wells (Leiby and Rubin, in press). Production of gasoline from these unconventional sources could have more adverse environmental effects than the conventional methods discussed in this article. However, the knowledge base concerning these impacts and their scales is still small.

Scales of the Crude Oil Extraction Processes

Petroleum fields may be small or extend across an entire region (Figs. 2a, 3). Extraction may begin within days of drilling a viable well, and a single oilfield may be tapped by multiple wells for a century or longer as different extraction technologies become available. Additional infrastructure including well pads, permanent roads, marine platforms and/or floating production vessels may be constructed to facilitate crude oil extraction over the lifetime of an oil field.

Oil wells produce for an average of 30 years (Miskimins 2009), but production varies over time. For instance, a typical well drilled in the Bakken Formation of North Dakota is projected to produce 550,000 barrels of oil during a 28-year lifespan, but over 65 % of this total volume is obtained within its first year, and a sharp decline in production volume follows (North Dakota 2011). As the original pressure of the reservoir declines, secondary and tertiary enhanced-oil-recovery techniques (e.g., water flooding, CO_2 injection) may be used to extend the productive life of the oil field. These measures often require construction of secondary infrastructure such as water wells, steam generation facilities, and water-injection bore holes.

Scales of Environmental Effects of Crude Oil Extraction

Most environmental effects of crude oil extraction are local in scale as they derive from drilling and pumping infrastructure installed at a particular site; however, some effects may extend across a landscape or region. Environmental effects of oil extraction (Table 1) include (1) alteration of subsurface pressure, potentially leading to seismic events or altered groundwater flow; (2) spills of hydrocarbons from well blowouts and conflicts, potentially resulting in fires on land or at sea; (3) land-cover changes from further infrastructure development; (4) competition for limited water resources; (5) the generation of toxic wastewater (process water or brine water) and effects of associated spills; (6) landscape changes, including subsidence and thermokarst formation; and, (7) post-production surface and subsurface contamination from abandoned or improperly decommissioned wells. The environmental effects of oil extraction range from local to global extent and may persist throughout geologic time (Fig. 2c).

Withdrawal of hydrocarbons from underground rocks may alter subsurface pressures and place stress on nearby faults; primary O&G production has been linked to seismic events in 38 locations globally (NRC 2012). Reservoirs rich in hydrocarbons but lacking in permeability may be purposely fractured with subterranean explosive charges and then propped open with rigid materials (e.g., sand or walnut shells) to increase flow. Subterranean fractures created through explosives or by steam can have large-scale effects on nearby communities by contaminating groundwater supplies and/or altering groundwater flow patterns (Manual 2010; Dittrick 2011; Kramer 2011).

Many environmental impacts are associated with oil spills, which may occur because of well blowouts during production or even because of resource-related conflict. The frequency of such oil spills and the magnitude of the associated impacts on surrounding ecosystems vary depending on concurrent environmental and social conditions specific to the region. Producing oil wells blow out at a rate of about one per 20,000 oil-well years and one per 2,500 oil-well-maintenance procedure operations (E&P Forum and UNEP IE 1997), a rate much lower than experienced during exploratory well drilling. Epstein and

Selber (2002) estimated that a total of 119–286 billion L of crude oil were unintentionally released into global waters and soils each year during oil extraction from the 1,776 land and 360 off-shore sites that they included in their analysis. As a comparison, it has been estimated that natural seepage of oil into the Gulf of Mexico amounts to approximately 19 billion L per year, a quantity large enough to produce oil slicks visible from space (Macdonald and others 1993). A recent environmental assessment of oil production in Ogoniland, Nigeria, found that terrestrial oil spills often cause fires that create a crust over the land and make remediation difficult (UNEP 2011). In that area of high rainfall, oil spills are quickly flushed to mangrove ecosystems that are critical for the maintenance of many aquatic species (UNEP 2011).

Armed conflicts related to oil may cause very large spills and fires. During the Persian Gulf War of 1990–1991, for example, an estimated 650 wells were set ablaze, thereby destroying approximately 442 million barrels of crude oil and releasing black smoke, sulfur dioxide, and nitrogen oxides into the atmosphere (Khordagui and Al-Ajmi 1993). Destruction of 751 oil wells led to groundwater and soil contamination, and up to 8 million barrels of crude oil were released into the ocean with detrimental effects on marine ecosystems that may last for decades (Hussain and Gondal 2008; Khordagui and Al-Ajmi 1993).

Infrastructure development for petroleum extraction also impacts terrestrial or marine life in a variety of ways. Producing wells affect vertebrates via both noise and traffic that extend beyond the well footprint (Lyon and Anderson 2003). Aboveground steam pipelines for extraction of bitumen from oil sands can fragment moose habitat (Dunne and Quinn 2009). In marine environments, bowhead whales (*Balaena mysticetus*) oriented away from sound levels consistent with those occurring 3–11 km from a drillship and dredging in the Canadian Beaufort Sea; the whales exhibited feeding cessation and call rate decreases (Richardson and others 1990). Oil infrastructure was identified as the principal cause for ecosystem fragmentation within the Laguna del Tigre National Park in Guatemala, where 90 % of documented human disturbances occurred within 2 km of petroleum roads and pipelines (CONAP 2006). Two years of Tahe oil field development in China's Taklimakan desert was found to be partly responsible for decreases in tree, shrub, and water cover and increases in desert, saline soil, and degraded grassland (Ouyang and others 2008). Construction of roads, borrow pits, and wells in the Brazilian Amazon caused siltation that had a detrimental impact on the density and taxonomic diversity of aquatic macro-invertebrates in nearby waterways (Couceiro and others 2010).

Re-vegetation and rehabilitation efforts at oil-field sites have had varied levels of success and sometimes unintended consequences. For example, several years following the abandonment of oil-field roads, the seeded roadbeds showed low plant diversity compared to the surrounding area, and non-native seeded species had spread into surrounding plant communities (Simmers and Galatowitsch 2010). Rehabilitation at Arctic oil-field sites has had mixed results, as have oilfield restoration efforts in Alberta, Canada (Jorgenson and Joyce 1994; Schneider 2002).

Produced water (also known as brine) is the largest byproduct associated with O&G production and was generated in the US at a rate of 210 million barrels per day in 1999 (Veil and others 2004; Khatib and Verbeek 2003). Produced water consists of salty water from the site mixed with water that may have been injected into the reservoir and can contain a mixture of oil, grease, dissolved organics, treatment chemicals, suspended solids, bacteria, metals, sulfates, and/or radioactive materials (Veil and others 2004). Oilfield wastewater-disposal facilities are typically large evaporation ponds ranging from 4,000 to 20,000 m^2 in size (Ramirez 2010). Bird fatalities in these ponds are generally attributed to oil, but sodium toxicity and surfactants have been implicated in a few cases (Ramirez 2010). Wastewater from O&G production may also be injected into disposal wells; a limited number of these wells have been shown to induce seismic events (NRC 2012). The US National Park Service has found that releases of produced water from O&G operations can create salt licks that affect behavior of black bear, elk, and other large mammals (NPS 2011).

Spills of produced water can be devastating to the local environment, but data on their spatial and temporal extents are sparse. More than 500 brine spills were reported in Louisiana between 1990 and the first half of 1998 (Bass 1999), and 900 brine spills per year were reported by the state of Oklahoma between 1993 and 2002 (Jager and others 2005). At one site in Oklahoma, the mean brine spill area was about 0.1 ha, and annual mean brine spill volumes were around 100 barrels (Jager and others 2005). Brine spills can cause underlying soil to become saline and denude the landscape of vegetation, leading to a "brine scar." The scarred soil is more susceptible to erosion, instigating enlargement of denuded areas for many decades (API 1997). In addition, components of terrestrial hydrocarbon spills can be locally phytotoxic (Efroymson and others 2004a), but, if applied, remediation treatments (e.g., fertilizer) can restore some native terrestrial vegetation within a few years (Efroymson and others 2004b).

Petroleum extraction may cause irreversible landscape changes such as subsidence in coastal regions. Subsidence from petroleum extraction has been documented along the coastlines of Louisiana, southern California, Venezuela, and The Netherlands as well as within the central portion of the North Sea (Morton and others 2006; Chilingar and Endres 2005; Nagel 2001). The effects of subsidence over a few square kilometers can be magnified and have serious consequences when the coastal wetlands and functional barrier islands are lost through inundation, as demonstrated along the coast of Louisiana (Morton and others 2006). Delayed effects are also possible; for instance, subsidence in eight hydrocarbon fields located in France, The Netherlands, Venezuela, and the North Sea occurred 1.6–13 years after the resource was depleted (Hettema and others 2002).

In Arctic regions, petroleum extraction may accelerate the melting of permafrost, thereby leading to landscape change through the formation of thermokarst (surface depressions that accumulate meltwater) and to the accelerated release of carbon currently stored within frozen tundra soils (Jorgenson and Joyce 1994; Walker and others 1987; Trucco and others 2012; Turetsky and others 2002). Cumulative thermokarst formation and flooding resulting from oilfield development in Arctic wetlands have been found to impact areas twice as large as the total area directly allocated to infrastructure (Walker and others 1987). These effects tend to lag infrastructure development by several years and may affect the landscape for up to several decades (Truett and Johnson 2000; Walker and others 1987). Because of low albedo and high thermal conductivity, water bodies formed from melting permafrost accelerate warming and melting around them; when coupled with a warming climate, this chain of effects has the potential to disintegrate an entire landscape (Walker and others 1987).

Permafrost soils in the northern hemisphere currently store about twice the amount of carbon as that contained in Earth's atmosphere (Trucco and others 2012). Therefore, as tundra soils melt they have the potential to release large volumes of CO_2 and methane into the atmosphere, leading to a positive feedback on global climate change (Turetsky and others 2002). In order to insulate the underlying ground ice, oil companies typically lay down up to 2 m of gravel under all roads and well pads (Streever 2002; Pelley 2001). This heavy use of gravel, typically obtained from surrounding stream beds, may lead to extensive quarried areas within oil fields [e.g., 22 % of Prudhoe Bay oilfield as of 1994 (Jorgenson and Joyce 1994; Walker and others 1987)]. Such raised gravel structures can function like dikes and prevent the flow of water during flood events, further altering the hydrodynamics of the surrounding Arctic landscape (Walker and others 1987). Because they resist re-vegetation, thick (often linear) gravel deposits also become problematic when rehabilitating Arctic landscapes following oilfield production (Streever 2002; Jorgenson and Joyce 1994).

Post-production well decommissioning is not an explicit step of the gasoline supply chain, but the environmental implications are unique to the production of gasoline and other petroleum products. A study of potential for below-ground CO_2 storage in North America found that the petroleum industry has left "many millions of exploration and production wells, most of which perforate otherwise intact caprock formations" (Nordbotten and others 2009, p. 743). Ideally, oil wells should be constructed with abandonment in mind so that the reserves and the freshwater aquifers penetrated by the wellbore are protected throughout geologic time, further surface pollution is prevented, and all regulatory requirements are met (Kelm and Faul 1999; Miskimins 2009). In practice, permanent protection is impossible because the materials used for well casings and cement plugs will eventually fail. A study of documented groundwater contamination incidents in the US state of Ohio found about one incident for every 180 O&G wells drilled during the 25-year study period; 22 % (41 out of 185) of these documented O&G-related incidents were related to leakage from orphaned wells (Kell 2011). The same study found that in Texas, there was one documented incidence of groundwater contamination out of every 890 O&G wells drilled over a 16-year period and that 14 % (30 out of 211) of these incidences were related to wells with no responsible owner, or "orphaned" wells (Kell 2011).

Decommissioning requirements for O&G wells vary widely across local jurisdictions and are often poorly enforced, although the situation in the US has certainly improved since the nineteenth century when wells were plugged with whatever materials were on hand (e.g., mud or tree stumps) (IOGCC 2009). There were an estimated 50,000 orphaned wells scattered across the US in 2008 (IOGCC 2009). A recent study of over 300 private O&G wells located in the 506 km^2 Big South Fork National River and Recreation Area, located in the US states of Tennessee and Kentucky, identified at least 45 orphaned wells and highlighted the difficulty of determining who will pay to remediate a contaminated water supply when the land, the below-ground mineral rights and the O&G extraction operations are all owned and/or managed by separate entities (NPS 2011). An Associated Press investigation following the Deepwater Horizon disaster (MSNBC 2010) found that about 50,000 O&G wells have been drilled in the US portion of the Gulf of Mexico and that approximately 27,000 of these wells have been abandoned with no monitoring for leaks. The cumulative effects of improperly abandoned wells on Earth's subsurface, including groundwater supplies, may be extensive and are likely to persist throughout geologic time (Miskimins 2009).

Decommissioning concerns also apply to the massive metal marine drilling and production platforms scattered throughout the oceans and seas. Since 2000, approximately 150 obsolete US drilling platforms have been decommissioned per year (National Commission 2011). Some defunct marine drilling platforms have been intentionally submerged off the coasts of Texas and Louisiana to form artificial reefs (National Commission 2011). A recent news report indicated that an "idle iron" policy has accelerated the rate of oil rig decommissioning in the Gulf of Mexico to as many as three per week and that groups of Gulf scientists, fishermen, and conservationists are expressing concern that the accelerated removal of the marine drilling infrastructure will destroy up to three acres of coral habitat per rig and impact as many as 30,000 fish per rig (Gates 2012).

Scales of Harvesting and Collecting Biomass

Obtaining the raw material for ethanol production involves harvesting or collecting feedstock, and handling, transporting, and storage of the material until it can be used at a biorefinery or preprocessed at an intermediate depot. Harvesting of energy crops may occur at decadal, annual, or seasonal time intervals depending on the feedstock and management system, giving this process a distinct temporal dimension from crude oil extraction. Soil management for the production phase can range from intensive annual tillage to systems with no tillage for a decade (e.g., perennial switchgrass) or more (e.g., woody crops) (Parrish and Fike 2005; USDOE 2011). Harvesting, collection, and storage techniques for energy crops will vary depending on the equipment available, social structure, expertise, and past experience of land managers.

Scales of Environmental Effects of Harvesting and Collecting Biomass

The cumulative effects of different crop harvesting and storage practices affect GHG emissions estimates for ethanol production. For example, modeling projections by Emery and Mosier (2012) indicate that net GHG reduction from producing ethanol rather than gasoline may change by as much as 11 % based solely on the dry matter loss emissions estimates derived from potential crop moisture levels analyzed in combination with different storage methods (e.g., ensiling vs. outdoor bales vs. indoor bales).

Biomass removal systems influence the extent of effects on local soil and water quality (Nelson and others 2004; Huggins and others 2011). Sediment erosion and transport into streams during rain events is a local process, but the associated impacts on aquatic biota depend on the stream distance travelled and the susceptibility of exposed biota and habitats. Headwater stream ecosystems are particularly affected by sedimentation if natural vegetation on slopes is replaced by cultivated row crops (Birkinshaw and Bathurst 2006). On the other hand, if barren, eroded or frequently burned slopes are planted with perennial bioenergy crops and managed to maintain groundcover, soil loss to streams will likely decrease. Tradeoffs must be considered as management practices may benefit some aspects of the environment while being detrimental to others. For instance, removal of wood residue from forests may reduce forest fire outbreaks but may also lead to increased erosion (Kocoloski and others 2011).

Step 3: Distribute Raw Material to Refineries

This first distribution step of the fuel cycle involves moving domestic and foreign crude oil to refineries via marine tanker, truck, and pipeline for conversion into gasoline and a series of co-products. For ethanol, this process step involves distributing domestic agricultural material to refineries, either directly or via depots, for conversion into ethanol and, potentially, co-products.

Scales of Distributing Raw Material for Gasoline

Crude oil moves through many landscapes including ice, sea, lakes, wetlands, barrier islands, Arctic environments, mangroves, prime farmland, and cities. Approximately half of the world's crude oil is transported by marine tanker (PetroStrategies 2011), and crude oil comprises more than 50 % of the mass of global marine cargoes (Burger 1997). Principal oceanic transport routes for crude oil run from the Middle East to Japan, from South America to Europe, and from Africa to the US (O'Rourke and Connolly 2003). In 2000, the Middle East exported 1,280 million t of oil to Asia, Europe, Australia, and the Americas (NRC 2003). About 80 million t of this oil arrived in the US Gulf of Mexico after travelling around the southern tip of Africa, a distance exceeding 18,820 km (NRC 2003). On average, four supertankers arrive in the US per day (GAO 2007; FTC 2004).

Crude oil is also transported to refineries across land and freshwater via pipeline, train, and truck. Domestic supplies from large oilfields are typically moved by pipeline, and the global network of oil pipelines is more extensive than the total length of railroads (Burger 1997). Sixteen billion barrels of oil were transported through the Trans-Alaska Pipeline System (TAPS) alone between 1977 and 2008, an amount sufficient to fill more than 19,000 tankers (APSC 2010). Built primarily on federal and state lands, the TAPS is about 1,300 km long (APSC 2010) and runs from Prudhoe Bay to Port Valdez, crossing three mountain ranges and 800 streams and rivers (APSC 2010). Constructing new pipelines is labor-intensive, and pipeline construction crews of as many as 1,500 people may make temporary footprints with their camps (APSC 2010). But the pipelines themselves can leave more permanent scars across the landscape (Schneider 2002).

Scales of Environmental Effects of Distributing Raw Material for Gasoline

Environmental effects from large marine tanker oil spills [e.g., 1989 Exxon Valdez (Table 3)] receive lots of media attention, but smaller marine-oil spills occur more frequently. Oil spills from tankers and ships in European marine transit routes typically extend between 0.01 and 100 km^2 (Redondo and Platonov 2009). Tankers may also discharge oil to oceans over time through poor operations or while rinsing out bilge, a corrosive mixture of water combined with cleaning agents, solvents, fuel, lubricating oils, and hydraulic oils that collects in the ship's hull (Körbahti and Artut 2010; Lucas and MacGregor 2006). Gas-chromatographic analysis of 2,343 oiled seabird corpses collected from Nova Scotia's Sable Island over 10 years indicated that 77 % of the 74 marine-oil discharge events responsible for the pollution were related to tanker cargo washings or slop tanks (Lucas and MacGregor 2006). The International Maritime Organization exacts fines for discharges of bilge water that exceed 15 ppm oil and grease, but these regulations only take effect at a distance of 22 km out from the nearest land; thus, in the absence of supplemental regulation, bilge wastes may be dumped closer to shore to avoid the treatment costs of meeting the 15 ppm limit (Körbahti and Artut 2010). Impacts of marine-oil spills on habitats and organisms can endure for years or decades (Mendelssohn and others 2012; Hussain and Gondal 2008).

Pipelines are susceptible to frequent rupture and spills across land and freshwater ecosystems (NTSB 2012). As pipelines age beyond 15 years, they may require more frequent maintenance to prevent potentially catastrophic spills, leaks, or explosions (Epstein and Selber 2002). A pipeline in Prudhoe Bay, Alaska ruptured in 2006 because of internal corrosion, spilling more than 1 million L of oil across a hectare of the North Slope's fragile tundra when it went undetected for 5 days (National Commission 2011, BBC News 2006). Based on 16 years-worth of data, the Alaska Department of Environmental Conservation (2012) reports that the majority of Alaskan crude spills are caused by pipeline corrosion; an average of 70 Alaskan spills each year release nearly 190,000 L of crude oil annually into the traversed environments.

The effects of oil spills from pipelines on terrestrial or freshwater environments vary in duration. A riverine fish assemblage exposed to diesel oil from a pipeline spill in Reedy River, South Carolina, was found to be similar to the reference group in just over 4 years (Kubach and others 2011), a result consistent with the relatively rapid recovery of freshwater ecosystems from other disturbances (Niemi and others 1990). In contrast, where sediments are anoxic and degradation of spilled hydrocarbons occurs over many decades (Wang and others 1998), biological recovery is expected to be slow. The recovery of marshes is often intermediate in duration; for example, some plant species along the Patuxent River of Maryland had recovered 7 years after fuel oil spilled from a ruptured pipeline, and some had not (Michel and others 2009).

Pipeline construction can involve clearing strips of land 15–30 m wide (Couceiro and others 2010) and may therefore adversely impact flora and fauna. However, management practices can reduce negative impacts from O&G activities or create opportunities to support positive effects. For example, in conjunction with the construction of the "Heavy Crude Pipeline" in Ecuador, a consortium of companies established a multi-million dollar fund (Eco-Fondo) to support biodiversity conservation (ten Kate and others 2004). This demonstrates how management decisions can influence the scale and direction of environmental effects and make assessments complex (e.g., with disturbances occurring in one spatial and temporal context while related conservation initiatives are being supported in others). Additional examples of management practices that could influence the scale of environmental effects include decisions (1) to use or rehabilitate previously disturbed sites, (2) to avoid environmentally sensitive areas (and/or contribute to their effective protection), and (3) to invest in monitoring and preventive maintenance.

Scales of Biomass Transport

Biomass is usually transported from the farm-gate to the refinery by truck, rail, or barge. In contrast to O&G distribution, distribution of material to ethanol biorefineries occurs on a local to regional scale concentrated around the "fuelsheds" where feedstock is sourced. Transporting bulky feedstock has been an obstacle to commercializing cellulosic ethanol (HEC and CABI 2010). The relatively low density of biomass per unit of energy creates challenges for economic distribution; therefore, feedstock and biofuel production currently tend to occur within the same region. Most ethanol plants in the US purchase grain from an area within 24 km from the plant (GTI 2010). The feasible biomass-production radius for a cellulosic biorefinery has been estimated to be 48 km (Mitchell and others 2008) or 80 km (Graham and others 2008), the latter assuming that farmers will not drive more than an hour to deliver their product. Decisions about whether to import feedstocks can also be influenced by the distance of biorefinery infrastructure from sea ports (Wellisch and others

2010). This localized scale of biomass distribution is partly due to the young age of the biofuel industry and the relatively higher cost of transporting bulky biomass. Trans-Atlantic transport of wood pellets from North America to European markets for biopower is expected to increase in the near term (Dwivedi and others 2011), so it is conceivable that supply chains for ethanol may also expand across the oceans in the future, depending on relative prices and policy incentives.

Scales of Environmental Effects of Biomass Transport

Transport of biomass can affect both air quality and GHG emissions. The level of impact depends on the distance travelled, mode of transport, and cumulative number of trips. Feedstock grown on farms throughout a large area could be transported to centralized preprocessing depots, processed into a standard form (e.g., pellets), stored, and shipped to refineries as needed, possibly returning animal feed to farms in the process (Eranki and Dale 2011). This uniform-format commodity-supply system has the potential to increase the efficiencies of biomass-handling logistics and transportation (Bals and Dale 2012). Ultimately, the economics of feedstock transport will vary with the price of the feedstock, gasoline prices, the feedstock energy content, transportation costs, exchange rates, policies (mandates or incentives), the biomass-to-biofuel conversion efficiency, and the biofuel selling price. So while a 40–80 km radius is a convenient estimate of the maximum feedstock transport distance, actual cellulosic supply chains could end up looking much different as all of these factors interact.

Step 4: Convert Raw Material into Fuel

The fourth step of the fuel supply chain involves the conversion of crude oil into gasoline through distillation and refining processes, and a combination of techniques (thermal, chemical, and biological) to convert biomass into ethanol. Manufacture of both fuel types requires inputs of water and produces solid and liquid waste streams. GHGs emitted during conversion of raw material into fuel have global effects for both gasoline and ethanol production.

Scales of Converting Oil into Gasoline

Petroleum refineries are typically large industrial complexes with extensive piping systems that are engineered to last for several decades. The world's largest oil refinery is the Reliance Jamnagar Complex in India, which produces over 190 million L of petroleum products per day and occupies more than 30 km^2 (Bechtel 2011). Large refineries are also found in Venezuela, South Korea, Singapore, Saudi Arabia, and the US. The largest US oil refinery is the ExxonMobil Refining & Supply Company facility at Baytown, TX, which produces 572,500 barrels of petroleum products per day (EIA 2009).

Gasoline production tends to be geographically concentrated. For instance, seven of the 10 largest US petroleum refineries are located in the Gulf Coast states of Texas and Louisiana, and these two states contain nearly 45 % of the nation's refinery capacity (EIA 2009, 2010). Crude oil is often transported to refineries from locations around the world. For instance, much of the petroleum processed by the US refineries arrives from Canada and Mexico by pipeline and tanker truck, and from the Middle East by ship.

Scales of Environmental Effects of Converting Oil into Gasoline

The major environmental effects of gasoline production are summarized in Table 1 and include emissions of gaseous, liquid, and solid waste streams from the long-term operation of oil-refining facilities and their associated infrastructure, some of which may be radioactively contaminated from the accumulation of naturally occurring radioactive material (Gray 1990). Although production of gasoline from unconventional sources is not included in this fuel comparison, the environmental impacts and GHGs associated with processing heavy oil and tar sands could be double or triple those associated with refining higher quality fuel (Karras 2010).

The spatial extent of air and water pollution from gasoline refining processes depends on the amount of crude oil refined, the processing technologies and control measures employed (Sorkin 1975), as well as wind and water flow. Air pollutants include volatile hydrocarbons, sulfur dioxide, nitrogen oxides, carbon monoxide, and particulate matter (Sorkin 1975) in addition to the CO_2 emissions. Unlike GHGs, which disperse globally, particulate emissions have local health impacts with the magnitude partially depending upon the population density near the refinery.

Hydrocarbon wastes from refineries are sometimes landfarmed, depending on contaminant concentrations, waste-disposal regulations, and land availability. If wastes are adequately diluted, bioremediation can be rapid, and the majority of the chemical load degrades within a few months to a few years (Maila and Cloete 2004). Nevertheless, multiple applications of hydrocarbon sludge may gradually increase the concentration of oil and grease if previous applications are not fully remediated. Long-term buildup of naturally occurring radioactive material in oil refineries may cause discarded equipment to necessitate management as radioactive waste (Tan and Pelletier 2009).

Scales of Converting Biomass into Biofuel

Like oil refineries, life spans of successful biorefineries will likely be several decades, especially for larger and more economical facilities. However, small biorefineries are more sensitive to commodity and oil prices, and sometimes have shorter lifespans (Krauss 2009). Biomass feedstock availability, conversion technology, policies, and market prices will largely determine the spatial and temporal extent of ethanol refining and production. Biofuel industries in Brazil and the US attained commercial scale in the 1980s (Keeney 2008; USEPA 2010b), and a large number of ethanol refineries were built in the US between 2004 and 2010 to meet RFS2 mandates (USEPA 2010b).

The extent and location of biorefinery siting is influenced by transportation networks, utility connections, and proximity to biofuel-demand centers and co-product markets (USDA 2010a). The scale of development of a regional or national collection of biorefineries is influenced by land suitable for feedstock production, as well as policies including tax incentives. As of October 2012, a total of 211 biorefineries were operating throughout the United States, with facility production capacities ranging from less than 4 million L/year to over 6,500 million L/year and a total capacity exceeding 51,800 million L/year (RFA 2012).

Commercial-scale biorefineries capable of processing cellulosic feedstocks do not yet exist in the US. Cellulosic facilities using agricultural residues will likely be located in arrays similar to existing biorefineries built for sugarcane ethanol (in south-central Brazil) and corn ethanol (in the Upper Midwest of the US). Those using woody feedstocks may co-locate with the pulp and paper industry and in areas where forest thinnings and residues are available. Thermochemical conversion processes that produce synthetic fuels may locate near existing petroleum refineries to take advantage of the extant distribution network.

Uncertainty about future policies and ethanol-supply-chain infrastructure compounds the difficulty of comparing the scales of the emerging biofuels industry to those of the evolving fossil-fuel industry. Four ethanol-supply-chain configurations (Fig. 4) can be envisioned depending on the density of biomass production and the capacity of the biorefineries that are constructed to convert biomass into ethanol [Richard (2010) identifies three of these]. Each of

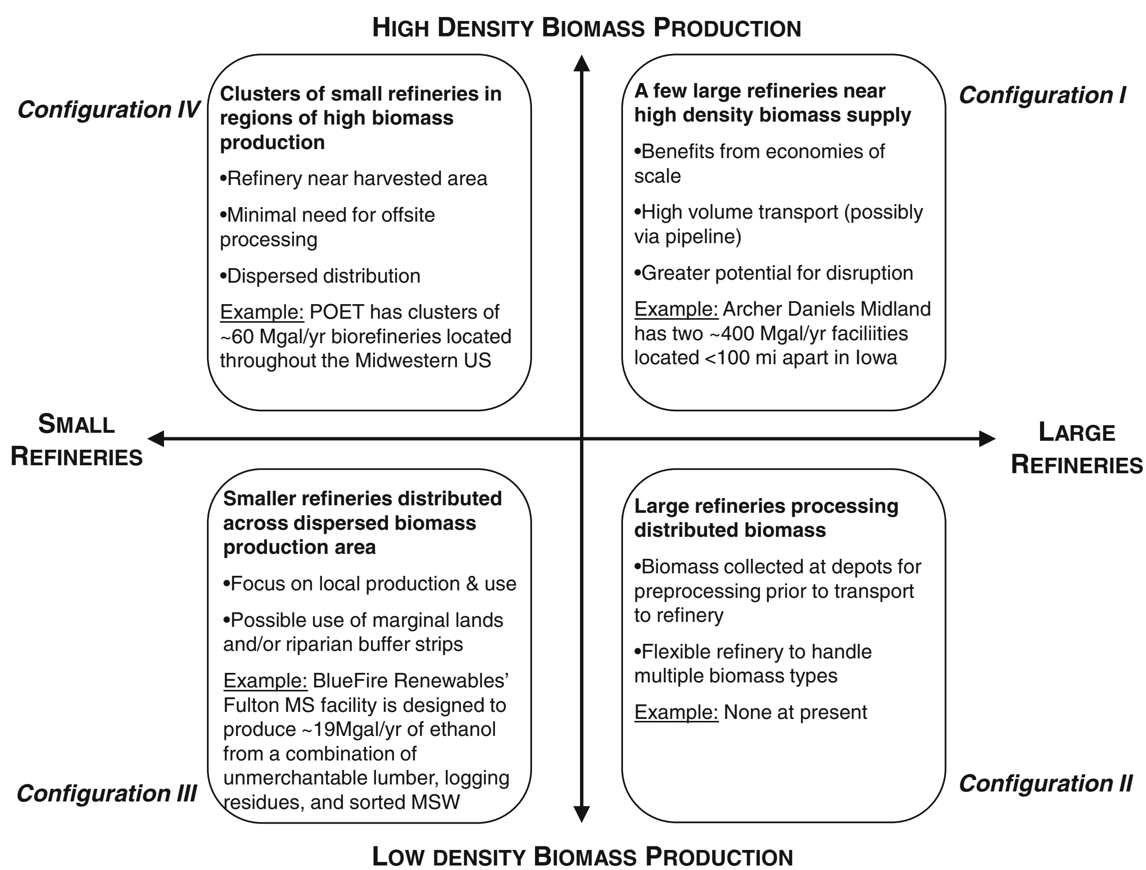


Fig. 4 Four potential ethanol-supply-chain configurations based on the density of biomass production (*vertical axis*) and refinery size (*horizontal axis*). Real-world examples are provided for three of the alternatives

the four configurations represents an approach to developing a biofuel chain that could be viable under certain conditions. Larger-capacity biorefineries may realize a lower unit cost of production than smaller biorefineries but require a larger supply of biomass to be delivered efficiently and a cost-effective distribution of the product and by-products to end users. Thus, a well-developed fuel-distribution system is important to the establishment of large biorefineries. Furthermore, feedstocks might come from spatially concentrated and intensive systems (e.g., large commercial farms with monocultures) or they might come from widely distributed and less intensively managed systems (e.g., residues gathered from several dispersed locations or production areas beneath utility lines). Distributed plantings could be supported by a preprocessing infrastructure that converts biomass into a commodity (e.g., pellets) that facilitates long-distance shipping (Hess and others 2009a).

The cellulosic ethanol-supply-chain configurations in Fig. 4 offer different opportunities and costs, suggesting that there may be an advantage to developing a heterogeneous supply-chain structure. These hypothetical supply chain alternatives can be compared to the established gasoline supply chain that exists primarily as a few very large refineries processing petroleum derived from widely distributed wells (most similar to Configuration II in Fig. 4).

Scales of Environmental Effects of Converting Biomass into Biofuel

Biofuel production is likely to have environmental effects of local extent that last from hours to decades (Table 2; Fig. 2d). Like their fossil-fuel counterparts, biorefineries are a source of criteria pollutants (i.e., particulates, ground-level ozone, carbon monoxide, sulfur oxides, nitrogen oxides, and lead that are regulated under the US Clean Air Act) and GHGs (Archer 2005; Wang and others 2007; Hess and others 2009b). Water use at a biorefinery can range from 3 to 6 L of water per liter of corn-grain ethanol produced, depending on facility type and age (Williams and others 2009; USEPA 2010b). Modeling analysis at a county resolution indicates that cellulosic-ethanol-biorefinery water consumption will vary by feedstock type and by region but is expected to range from two to 139 L of water consumed for each liter of ethanol produced (Williams and others 2009; USEPA 2010b; Chiu and Wu 2012). Effects of water withdrawals for biofuels production are a function of the location (including competing human uses) and conversion process. Many ethanol plants currently use groundwater to ensure quality, so competition for groundwater may limit production in the future (Scown and others 2011; Wu and others 2009). Wastewater discharges from biorefineries are variable and depend on production processes and plant-specific control technologies (USEPA 2010b).

Steps 5 & 6: Distribute and Use Fuel for Transportation

As previously discussed in the "Methods Used for Fuel Supply Chain Comparison" section, the steps of distributing fuel and using it for transportation involve similar substeps for gasoline and ethanol and have potential environmental effects with generally similar scales (Tables 1, 2). However, while the particular effects of a gasoline or ethanol spill during transportation are likely to be similar in extent for similar fuel volumes at particular locations (e.g., pollution of underlying soils and groundwater and phytotoxicity), the actual extent and effects of gasoline spills are distinct due in part to the wider spatial distribution of production, longer transport distances, and larger scales of operation relative to ethanol.

Scales of the Processes of Fuel Distribution and End Use

Scales of distribution of gasoline depend on the relationship between supply and demand, as well as transportation costs. Countries and regions that do not have sufficient refining capacity to meet local demand import some gasoline. Europe has an increasing gasoline surplus and needs to dispose of that surplus (Purvin & Gertz Inc. 2008). The three distinct Canadian regions (western Canada, Ontario, and Quebec, and the Atlantic coast) tend to be self-sufficient with respect to using gasoline refined within the region (Natural Resources Canada 2009). Most US gasoline is transported from the Gulf Coast refineries to the portions of the country that lie east of the Rocky Mountains. Gasoline is also distributed from refineries located along the East Coast and in the Midwest. California's gasoline is produced almost completely within the state to meet higher state standards. Thus, distributing gasoline for end use is a process that mostly ranges from landscape to continental spatial extent. However, surplus gasoline has been exported to the US from Europe, and new Middle Eastern and Indian refiners are targeting the US for gasoline sales (Purvin & Gertz Inc. 2008). European refiners will also likely increase sales of gasoline to Africa and the Middle East (Purvin & Gertz Inc. 2008). Therefore, the maximum scale of distribution of gasoline is moving toward global scale.

US ethanol, on the other hand, is primarily transported outward from the Midwest, where six states are currently responsible for nearly 75 % of total production (USEPA 2010b). The ethanol transportation and distribution infrastructure radiates out from the center of the country to storage facilities and petroleum blending terminals near major population centers. Relatively small amounts of ethanol are imported by the US and arrive at ports on both coasts, primarily from Brazil and the Caribbean Basin Initiative countries (USEPA 2010b; RFA 2010) although in

2011, the US was a net exporter of ethanol to Brazil (EIA 2012). Changes in relative prices, exchange rates, tariffs, and subsidies influence the flow and volume of ethanol trade among Brazil, the US and other nations. Ethanol distribution lacks a dedicated pipeline network and, because of both geographic and chemical incompatibility concerns, is unable to make extensive use of the existing petroleum infrastructure. Most ethanol is transported from refineries to storage and blending terminals by rail and tanker trucks, and the remainder moves by barge (USDA 2007).

Gasoline and ethanol delivery infrastructures merge at the petroleum-blending terminals located in or near major metropolitan regions and serviced by petroleum-product pipelines. Of the 1,063 US gasoline terminals, nearly 500 have ethanol-storage facilities, but only 88 have direct rail service (USEPA 2010b; USDA 2010b). The remaining gasoline terminals receive ethanol via tank trucks that shuttle the fuel from rail yards and barge terminals (USDA 2010b). Total ethanol-storage-tank capacity has grown from about $\sim$41 million L in 2000 to more than 111 million L in 2012, while gasoline storage has steadily declined during the same time period, dropping from $\sim$13 to $\sim$9.5 billion L (EIA 2010). At the petroleum terminal, ethanol is blended with gasoline (currently up to 10 % by volume to create E10) or is distributed directly to retail outlets for onsite storage and eventual mixing with gasoline for sale as E85 (up to 85 % denatured ethanol by volume). Although E10 is now found throughout the US, fewer than two percent of US fueling stations were equipped to dispense E85 or biodiesel in 2009 (USDOE 2010). Most E-85 stations are in Midwestern areas near sources of production (as shown by the live map at (http://e85prices.com/e85map.php). In contrast, Brazil's gasoline is blended to about 25 % ethanol, and the blend rate can be adjusted in response to markets.

Scales of the Environmental Effects of Fuel Distribution and End Use

Leaks or accidental spills of fuel during transportation to end users are likely to occur in different locations for ethanol and gasoline. Air and water pollution are the main environmental concerns associated with liquid fuel transportation and distribution. USEPA criteria pollutants, GHGs, and toxic chemicals associated with fuel transportation are lower for pipelines than other options (USEPA 2010b). The atmospheric lifetime of these pollutants ranges from days to centuries, and the spatial scale of activity ranges from local deposition to global transport.

Local effects from fuel spills are a major environmental impact, and their accumulated effects may be quite large. Bulk fuel terminals are a common location of urban spills of petroleum products (Li and McAteer 2000). Repeated urban spills have cumulative environmental impacts, including direct toxicity to aquatic and terrestrial plants and animals, loss of soil, and freshwater quality because of stimulation of microbial and algal populations and groundwater pollution (Li and McAteer 2000). Water quality can also be impacted by accidental spills and leaks from underground storage tanks (USTs) for fuel. The potential impacts of leaking USTs on groundwater are a concern in the US, where the majority of approximately 5,000 USTs store petroleum or petroleum-based products (USEPA 2012). Ethanol blends have a greater potential to corrode the materials traditionally used to store gasoline both above- and below-ground (Niven 2005). Ethanol degrades relatively quickly, but it can retard the degradation of benzene, toluene and xylene by changing the geochemistry of the surrounding soil (USEPA 2010b). Compared to spills of gasoline alone, plumes of gasoline mixed with ethanol may have greater or longer-term effects on drinking water resources (USEPA 2009a, b; Powers and others 2001; Ruiz-Aguilar and others 2002).

Air pollution from fuel combustion in vehicles is the primary environmental effect of end use. Many of those pollutants are GHGs and readily become well mixed in the atmosphere at the global extent; others, such as particulate emissions, are regional in scale. Although CO_2 emissions from liquid transportation fuel combustion may appear inconsequential at local and short-term scales, they have global consequences in the form of climate change effects over centuries (IPCC 2007). A combination of innovations and environmental regulations has enabled the production of automobiles that emit $\leq$1 % of the mass of air pollutants than they did 40 years ago (Greene 2010). Nonetheless, the transportation sector currently accounts for more than 70 % of global carbon monoxide emissions and 19 % of global CO_2 emissions (Balat 2011). When compared to gasoline emissions, a flexible-fuel vehicle using E85 may reduce nitrogen oxides and carbon monoxide but increase formaldehyde and acetaldehyde emissions (Yanowitz and McCormick 2009).

Discussion of Key Scale Differences and Similarities

Several key scale differences in fuel-production processes and environmental effects can be discerned from the Stommel diagrams (Fig. 2). Overall, the steps for the gasoline supply chain (Fig. 2a) are often more extensive than those for ethanol (Fig. 2b). Petroleum exploration and extraction (Steps 1 and 2) occur across every continent and ocean; large volumes of crude oil are shipped across the seas (Step 3); and the collection, refining, and distribution phases of gasoline (Steps 4–6) may occur in distinct regions that are far apart. In contrast, although ethanol-supply chains are present on most continents, they tend to occur within a single landscape or region, mostly because of economic limitations to long-distance transport of biomass.

The extent and location of future disturbances associated with fuel supply chains are uncertain since evolving technology may enable new pathways for ethanol and already allows petroleum extraction from sources that were previously considered inaccessible or uneconomical (e.g., sedimentary basins residing deep beneath Earth's oceans). Ultimately, the cumulative spatial extent of biomass feedstock establishment (Step 1) is limited by the availability of locations with favorable soils and climate (i.e., arable land). Oil exploration has the constraint of suspected petroleum availability, but it can extend to more remote locations than feedstock establishment and can occur in aquatic and non-arable areas (Fig. 2a, b).

The Stommel diagrams indicate that the environmental effects of extraction (Step 2) and distribution (Step 3) associated with gasoline production have a larger maximum spatial extent than those of ethanol (Fig. 2c, d). The difference in maximum extents of environmental effects is influenced by the greater number of locations where oil can be found compared to where bioenergy feedstocks can be produced. In addition, oil extraction has extensive and long-lasting effects across subterranean resources, whereas the effects of feedstock harvest and collection are generally limited to surface resources and shorter timescales.

The extents of effects from exploration and feedstock production (Step 1) and from refinement (Step 4) appear similar for the two fuels (Fig. 2c, d) although the type and location of these effects are distinct. Exploration for oil can involve seismic surveys, drilling and well logging, deployment of marine platforms, and infrastructure construction (such as roads, bridges, work camps, and air fields) that have regional impacts (Table 1). Establishment of biomass feedstock entails planting the energy crop or identifying available residues or wastes. Land management associated with bioenergy establishment can have regional effects on water quality and hypoxia, but those effects may be positive if perennial crops and proper management practices are employed (Table 1). Environmental effects of converting oil into gasoline occupy similar spatial extents as effects of converting biomass into ethanol, for both involve alteration of chemical and physical properties and occur in production facilities that generate local or regional air and/or water pollution.

The duration of environmental effects of gasoline exploration, extraction, and production exceed those for ethanol, but the duration of distribution effects are similar for the two fuels (Fig. 2c, d). Oil exploration, extraction, and production involve processes that can have long-term or irreversible impacts such as subsidence, establishment of infrastructure in pristine areas, alteration of ground water flows, and surface and subsurface contamination (Table 1). In contrast, the duration of environmental effects of establishing and harvesting or collecting biomass for ethanol occurs on the order of years to decades.

A critical temporal distinction exists when comparing ethanol and gasoline life-cycles. Oil deposits were established millions of years in the past. The use of oil transfers into today's atmosphere GHGs that had been sequestered and secured for millennia and would have remained out of Earth's atmosphere if not for human intervention. While the production and use of bioenergy also releases GHGs, there is an intrinsic difference between the two fuels, for GHG emissions associated with biofuels occur at temporal scales that would occur naturally, with or without human intervention. The cycle of sequestration and release of carbon and nutrients as plants grow, die and decay occurs on the order of years to decades with or without the implementation of a bioenergy system. Hence, a bioenergy cycle can be managed while maintaining atmospheric conditions similar to those that allowed humans to evolve and thrive on Earth. In contrast, massive release of fossil-fuel carbon alters this balance, and the resulting changes to atmospheric concentrations of GHGs will impact Earth's climate for eons (IPCC 2007).

Both gasoline and ethanol production have the potential to emit pollutants to the air, water, and land during multiple process steps. The US oil and gas industry generates more solid and liquid waste than municipal, agricultural, mining, and other industrial sources combined (O'Rourke and Connolly 2003). There is no comparable estimate for the ethanol industry, which currently operates at much smaller scales than gasoline. Each fuel-production pathway has the potential to pollute surface water resources at a regional scale, either through nutrient and sedimentation runoff during biomass feedstock establishment and management or through aquatic oil spills during exploration, extraction, and transportation of crude oil (e.g., Table 3, UNEP 2011). Water quality and hypoxic conditions change year to year depending largely on precipitation patterns and oceanic currents (Dale and others 2010b). Although effects from oil spills may only last for years or decades (Lin and Mendelssohn 2012), the cumulative effects of improperly abandoned oil wells and fractured rock formations have the potential to lead to centuries of groundwater contamination (Miskimins 2009).

As ethanol-production technologies become standardized and research on the effects of these technologies matures, the bounds of the Stommel diagrams for ethanol (Fig. 2b, d) will become more precise. However, because of the potential for catastrophic accidents for materials under pressure in oil wells, as well as the hazards associated with shipping large quantities of liquid petroleum products, the environmental effects of gasoline at different spatial and temporal scales will continue to have a high degree of uncertainty.

Factors Complicating Scale Comparison

In addition to inherent uncertainty, comparison of environmental effects of ethanol and gasoline production across different scales proves challenging for several reasons:

(1) Petroleum and biofuel systems are dramatically and qualitatively different throughout the supply chain. Analogous supply-chain comparisons are inherently limited by fundamental differences between the two fuel sources, such as the need to extract a nonrenewable resource from a subsurface geologic formation versus the capability to grow and harvest a constantly regenerating crop on Earth's surface. There is no way to put some effects into quantitatively comparable terms (e.g., the effect of permanently depleting subsurface deposits of petroleum).

(2) While the scales of some environmental effects are relatively easy to measure (e.g., direct land footprint or average water consumption of a process), it is difficult to attribute other environmental effects (e.g., changes in water quality and air quality, land-use change) to energy production. This difficulty in attribution is especially problematic when evaluating future feedstock development scenarios since many bioenergy crops and residues have potential for multiple end uses (e.g., food and fiber) and coproducts.

(3) Management decisions and their related environmental effects throughout both supply chains depend on the systems' environmental, economic, and policy contexts (Efroymson and others, this issue). Given that nearly all arable land is affected by human activities and that the impacts of management practices depend on local context, it is difficult to make projections about specific effects based on average and aggregated data for generalized pathways. The effects always depend on interactions among many local factors that may not be fully understood, and erroneous conclusions about sustainability can be drawn when information is only pertinent to particular times and places (Turner and others 2001). For example, a life-cycle analysis might conclude that producing a given unit of fuel requires the disturbance of 1 ha of land, but effects of this disturbance depend on prior uses of that land and whether it is isolated from other disturbances, or part of a road or an extensive seismic-line network.

(4) The effects of either fuel-production pathway are strongly influenced by management practices and decisions. Environmentally sound planning and responsible management can avoid or mitigate several impacts discussed, or amplify them. In many cases, insightful management can contribute to converting potentially negative impacts into positive effects [e.g., by utilizing and rehabilitating degraded resources or establishing biodiversity "offsets" (ten Kate and others 2004)]. Management practice combined with contextual issues (prior point) make it difficult to reach broad conclusions about effects that will be applicable in every situation.

(5) Land-use changes resulting from energy production have various degrees of reversibility (Dale and others 2011a) that are not captured by Stommel diagrams. How does one compare the loss of a unit of marshland along the Gulf coast to subsidence (a permanent loss of land to the sea) with the use of a unit of prairie grassland for a bioenergy crop? Land dedicated to bioenergy crop production can be either replanted with alternative vegetation almost immediately or taken out of feedstock production without any significant change in functionality. By contrast, some land disturbance effects of petroleum production may only be reversed through years of restoration, and subsurface disturbances may persist throughout geologic time.

(6) Understanding ways that biofuel production might affect the environment over space and time necessitates comparing the effects of the proposed activity to conditions that might exist in the absence of the proposed activity (i.e., continued production of gasoline). However, characterizations of business-as-usual conditions and projections of future energy production processes inevitably rely upon assumptions and modeling that are inherently limited. Many siting decisions concerning preferred feedstocks, biorefinery capacities, and associated infrastructure have yet to be made, particularly for cellulosic ethanol production. Because the commercial biofuels industry is in its infancy, nearly all large-scale future bioenergy systems must be simulated to estimate their potential large-scale environmental effects. Preferred technologies and best management practices for gasoline production also continue to evolve and improve. Fuel production targets remain in flux as policy and global economic conditions change. Researchers must be careful not to project effects of future fuel production based on past practices when future material management and market conditions are expected to be different.

(7) As human population and affluence continue to rise, the scale of energy use and magnitude of GHG

emissions will push substantially upward (Rosa and Dietz 2012).

Conclusions

Producing and using energy consumes resources and has environmental impacts. Although both gasoline and ethanol production may result in negative environmental effects, this study indicates that ethanol production traced through a supply chain may impact less area and result in more easily reversed effects of a shorter duration than gasoline production. Effects of the gasoline pathway have distinctive spatial extents involving remote and fragile ecosystems, the significant subterranean dimension of disturbances, and the temporal shifting of huge volumes of GHGs from prehistoric times to today's atmosphere. Ethanol expansion has the potential to reduce environmental impacts when compared to current gasoline production and its support systems, but research, monitoring, and enforcement are needed to guide choices toward more sustainable resource management. Indeed, there is potential for combined environmental and social benefits from careful landscape design of bioenergy cropping systems (IEA 2011; Parish and others 2012).

A variety of energy pathways are possible over the coming decades, and each will lead to a different cumulative extent and duration of environmental impacts. The International Energy Agency (IEA 2011) projects that biofuels will be the second largest contributor to the portfolio of technologies needed to reduce transportation fuel emissions to levels necessary to achieve 50 % reduction in energy-related in CO_2 emissions by 2050 (as compared to 2005). Under this IEA BLUE Map scenario (2011), biofuels are expected to increase from 2 to 27 % of the global transportation fuel supply by the year 2050. Under the same scenario, gasoline is projected to drop to 13 % of the global transportation fuel supply by 2050 (IEA 2012). Given the pressing need for alternatives to fossil-fuel sources, commercial biofuel production may expand before sufficient relevant research can be completed and the most appropriate policies determined and implemented. The potential expansion of biofuels production makes it imperative for leaders and decision makers to promote an adaptive-management approach (Walters and Hilborn 1978) that fosters the incorporation of new information about bioenergy cropping systems simultaneously with expanding their use (Dale and others 2010c).

This analysis is a critical first step toward understanding the environmental scale of sustainability of gasoline and ethanol production and suggests development of a complementary multi-scale analysis of socioeconomic effects (Dale and others, in press), which are also likely to operate at several spatial and temporal scales. Measuring, modeling, and analyzing environmental and socioeconomic effects at different scales and using the results to plan and implement a sustainable liquid fuel supply chain require a concerted interdisciplinary effort. We therefore recommend that more interdisciplinary research be supported and that frameworks be developed for assessing impacts across the supply chain and at different scales.

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