SUSTAINABLE TRANSPORTATION ENERGY PATHWAYS

A Research Summary for Decision Makers

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Chapter 7: Comparing Land, Water, and Materials Impacts

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The environmental impact of transportation fuels and vehicles doesn't stop at GHG emissions but also includes impacts on land, water, and materials used in their production. Local land-use impacts occur where biofuel feedstocks are grown; these must be acknowledged and weighed against the land-use impacts of oil production. (Note that in addition to its direct local impacts, biofuel production can have important indirect impacts; these are considered in Chapter 12.) Production of fossil fuels, biofuels, electricity, and hydrogen all have water footprints that must be considered in any comprehensive assessment of environmental impacts. In addition, advanced vehicle technologies use materials that might become a barrier to development if they are either scarce or else concentrated in a few countries. This chapter focuses on work that has been done so far comparing the sustainability of different fuel/vehicle pathways along these lines.

Local Land-Use Impacts of Transportation Fuel Production

Government support of major biofuel programs in the United States and other countries has intensified discussion of the land-use implications of biofuels, among other impacts.[1] However, our understanding and measurement of these impacts are at present rather limited. We do know that any land disturbance caused by fuel production, whether of biofuels or oil, not only has an impact on the ecological integrity of the land and its wildlife but also results in GHG emissions. Here we compare the local land-use impacts of biofuel and oil production.

Local land-use impacts of biofuel production

Recent studies point out that if biofuels are produced on carbon-rich lands such as forest or tropical peatlands, this can release large amounts of greenhouse gases that may take decades of biofuel production to sequester back.[2] Land-use impacts from biofuel production can also occur farther afield due to the global reach of commodity markets; this topic—indirect land-use change—is taken up in Chapter 12.

Aside from the question of emissions, the use of monocultural feedstocks (such as corn) to make biofuels can reduce biological diversity and the associated biocontrol services in agricultural landscapes. A simple land-use intensity metric (such as acres per energy unit of fuel produced) is not a good indicator of these impacts, in part because it does not reflect the impact of the land use on habitat integrity, wildlife corridors, and interactions at the edges of the affected area.

By any of these measures, biofuels made from agricultural crops can severely degrade natural habitats. To mitigate these effects, monocultures should be replaced by "natural, diversified and multifunctional vegetation that could meet the broad demand for goods and other resource functions in a sustainable fashion."[3]

Biofuel-crop harvesting practices can affect soil erosion and the nutrient and organic content of the soil, which in turn can affect the use of fertilizer. For example, if crop residues are removed from the field and used as a source of energy in the production of a biofuel, soil erosion might increase and fewer nutrients and less organic matter might be returned to the soil. Additional fertilizer might be required to balance any loss, and the use of additional fertilizer will result in additional environmental impacts.

Land-use impacts of oil development

Many studies examining the land-use impacts of oil and gas production have found significant levels of habitat loss, fragmentation, and other ecological impacts associated with these developments.[4] Yeh et al. were the first to study GHG emissions associated with the land disturbance caused by oil production.[5]

Using oil wells in California and Alberta as examples of conventional oil production, and oil sands production in Alberta as an example of unconventional oil production, Yeh et al. found that the land-use impacts of oil production in Canada can be substantial, as it disturbs large tracts of land in the boreal region. Since a large portion of the disturbed area is on peatlands (a special formation of soil that slowly accumulates carbon over thousands of years and stores ten times more carbon than regular soil found in most places), the carbon emissions can be quite high.

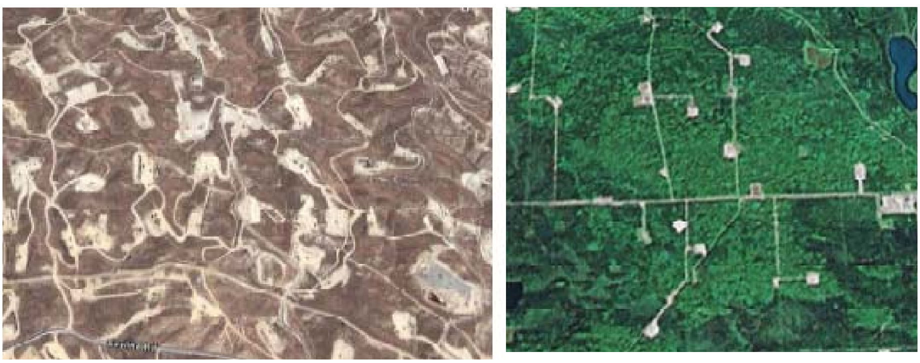
Conventional oil development causes land disturbance when infrastructure such as well pads, pipelines, and access roads are installed, and when seismic surveys are done. Typically, few oil wells are drilled during exploration. During development, well density increases until oil production rates drop below economically recoverable levels. Wells are shut in and abandoned afterward. In Canada, oil wells need to be reclaimed and certified to ensure that abandoned wells have a land capability that is equivalent to predrilling conditions, though the compliance rate has been declining since 2000.

Oil sands projects are generally located in northeast Alberta, with some development extending to the northwest of the province in the Peace River region and east into Saskatchewan, an area classified as boreal forest. Bitumen is extracted from oil sands using in situ recovery or surface mining. In situ recovery involves drilling wells into deposits typically deeper than 100m and injecting steam into the reservoir, reducing the bitumen's viscosity and allowing it to be pumped to the surface. Infrastructure such as central processing facilities and networks of seismic lines, roads, pipelines, and well pads must be built to support in situ recovery.

Surface mining of bitumen, used for shallower deposits, requires the clearing and excavation of a large area; it involves draining peatlands, clearing vegetation, and removing peat, with subsoil and overburden being removed and stored separately. The total land disturbance includes a mine site, overburden storage, and tailing ponds. Disturbed peat is stockpiled and stored until reclamation, when it may be used as a soil amendment. The drained and/or extracted peat will begin to decompose, releasing a combination of CO_2 and CH_4 (methane) depending on peat moisture conditions.[6] When the functional vegetation layer at the surface of a peatland is removed, the disturbed ecosystem loses its ability to sequester CO_2 from the atmosphere, so foregone sequestration must also be factored in. Reclamation of surface mines typically involves reconstructing self-sustaining hydrology and geomorphology on the landscape.[7] A mixture of peat and soil from the original lease and surrounding sites is used to cover the end substrates. The landscape is subsequently seeded and revegetated.

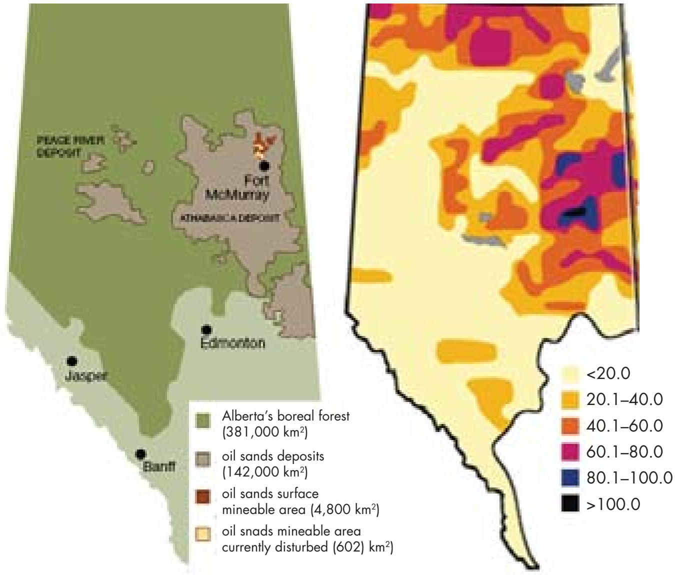
Yeh et al. calculated the amount of land disturbed per unit of fuel produced for both historical and current production of conventional oil in California and Alberta. They used image analysis to determine the land area disturbed per well, dividing the total disturbed area by the number of distinguishable well pads counted in each image. They found that the land area disturbed per well is almost three times larger in Alberta than in California, averaging 1.1 hectares per well (ha/well) in California compared with 3.3 ha/well in Alberta. As a result, the energy yields (PJ per ha of disturbed land) are roughly two times higher for California oil production compared with Alberta conventional oil production. In both places, oil production peaked around 1985 and has been declining ever since. Thus the marginal land-use impact of oil production has increased, with more land disturbance and less energy output.

IMAGES OF LAND DISTURBANCE FROM FOSSIL FUEL PRODUCTION



These images—extracted from Google Earth and attributed to Telemetrics, TeleAtlas and Digital Globe 2009—show the land disturbance resulting from oil production in Elk Hills, California (left), and Alberta (right).

LOCATION OF OIL SANDS DEPOSITS AND PEATLANDS CARBON DEPOSITS IN ALBERTA'S BOREAL REGION



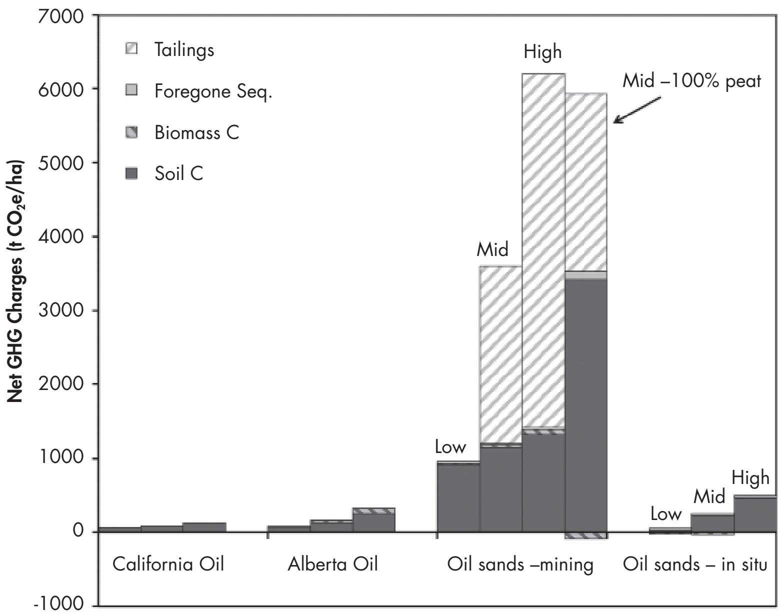
Alberta's conventional and oil sands deposits (left figure) sit right on top of one of the world's largest carbon deposits (right figure, contour interval is 20 kg•m2). Source for the figure on the left: Government of Alberta, Alberta's Oil Sands, Reclamation, http://www.oilsands.alberta.ca/reclamation.html#JM-OilSandsArea. Source for the figure on the right: D. H. Vitt, L. A. Halsey, I. E. Bauer, and C. Campbell, "Spatial and Temporal Trends in Carbon Storage of Peatlands of Continental Western Canada Through the Holocene," Canadian Journal of Earth Sciences *(2000): 683–93. Boreal peatlands store 85 percent of global peat and contain about six times more carbon than tropical peatlands. Yeh et al. estimate that 15 percent of conventional oil and 23 percent of oil sands development occurs in peatland.*

In addition to having large environmental and ecological impacts, land disturbance also contributes to GHG emissions. Natural carbon stocks increase and decrease as a result of land disturbance through a variety of mechanisms. The mechanisms Yeh et al. examined include clearing of vegetation, loss of soil carbon, foregone sequestration, and resequestration due to reclamation and/or forest regrowth. They also assessed CH_4 emissions from tailings ponds and peat stockpiled during oil sands surface mining operations. Though CH_4 emissions from tailings ponds are different from biological carbon typically included in land-use analysis, these emissions were included because of the large land areas covered by tailings ponds, the high CH_4 emissions, and the extent to which emissions can be affected by mitigation decisions related to land-use management.

Peatland conversion and tailing ponds are the largest sources of GHG emissions of oil production examined in the study. As Canadian oil sands production may reach 1.5 billion barrels per year in 2030[8], this may result in an additional 50,000–96,000 hectares of cumulative land disturbance and 47–580 megatonnes of CO_2e emissions resulting from surface mining between

2010 and 2025; in situ production may add 9,100–21,000 hectares of land disturbance and 0.1–10 megatonnes of CO_2e emissions during the same period (not including upstream disturbance from the use of natural gas). These findings emphasize the importance of restoration activities after oil sands production has been completed, not only to reduce land-related CO_2 emissions but more importantly to recover ecological landscapes and sustain high biodiversity, hydrologic cycles, and forest ecosystems.

NET GHG CHANGES OVER 150 YEARS FROM LAND DISTURBED BY OIL PRODUCTION



Yeh et al. quantified changes in carbon stock and CH_4 emissions per unit of area disturbed by conventional oil production and oil sands over a modeling period of 150 years, assuming reclamation back to a natural state after project completion. Oil sands surface mining is far and away the largest contributor on this score. Source: S. Yeh, S. M. Jordaan, A. M. Brandt, M. R. Turetsky, S. Spatari, and D. W. Keith, "Land Use Greenhouse Gas Emissions from Conventional Oil Production and Oil Sands," Environmental Science and Technology, *(2010): 8766–8772.*

Comparing the land-use GHG impact of oil and biofuels

Three important variables determine the direct land-use greenhouse gas (GHG) impact of liquid transportation fuels:

- energy yield—that is, the amount of energy produced per unit of land disturbed
- GHG emissions produced per unit of land disturbed
- GHG emissions produced per unit of energy output

When we compare the land disturbance from fossil fuel and biofuel production, it is the energy yield that greatly distinguishes the two. Due to the significantly lower energy output per unit of land used for crop production versus fossil energy production, biofuels require orders of magnitude more land than do petroleum fuels for the same amount of energy produced. Thus, although GHG emissions per unit of land disturbed by oil production can be comparable to or higher than emissions from biofuel production, land-use GHG emissions per unit of energy output for oil can be significantly lower than for biofuels.

COMPARISON OF DIRECT LAND-USE IMPACTS, BIOFUEL VS FOSSIL FUEL PRODUCTION

Energy Source		Energy Yield (PJ/ha)	GHG Emissions (t CO2e) per Hectare	GHG Emissions (g CO2e) per MJ
Fossil fuel				
California oil	historical impacts marginal impacts	0.79 (0.48–2.6) 0.55 (0.33–1.8)	73 (59–117)	0.09 (0.02–0.25) 0.13 (0.03–0.35)
Alberta oil	historical impacts marginal impacts	0.33 (0.16-0.69) 0.20 (0.092-0.40)	157 (74-313)	0.47 (0.12-1.98) 0.78 (0.20-3.39)
oil sands—surface mining		0.92 (0.61-1.2)	3596 (953-6201)	3.9 (0.83-10.24)
oil sands - in situ		3.3 (2.2-5.1)	205 (23-495)	0.04 (0.0-0.23)
Biofuel				
palm biodiesel (Indonesia/Malaysia)	tropical rainforest	0.0062	702 +/– 183	113 +/– 30
palm biodiesel (Indonesia/Malaysia)	peatland rainforest	0.0062	3452 +/– 1294	557 +/– 209
soybean biodiesel (Brazil)	tropical rainforest	0.0009	737 +/– 75	819 +/– 83
sugar cane (Brazil)	cerrado wooded	0.0059	165 +/– 58	28 +/– 10
soybean biodiesel (Brazil)	cerrado grassland	0.0009	85 +/– 42	94 +/– 47
corn ethanol (US)	central grassland	0.0038	134 +/– 33	35 +/– 9
corn ethanol (US)	abandoned cropland	0.0038	69 +/– 24	18 +/– 6

Note that values for fossil fuel are single estimates consisting of the mid-range values; the upper-bound and lower-bound estimates are reported in parentheses. Values for biofuels include standard deviations. Source: S. Yeh et al., "Land Use Greenhouse Gas Emissions from Conventional Oil Production and Oil Sands." Biofuel estimates are based on data from J. Fargoine, J. Hill, D. Tilman, S. Polasky, and P. Hawthorne, "Land Clearing and the Biofuel Carbon Debt," Science *319 (2008): 1235–38, which assumes a 50-year biofuel production period.*

Water Resource Impacts of Transportation Fuel Production

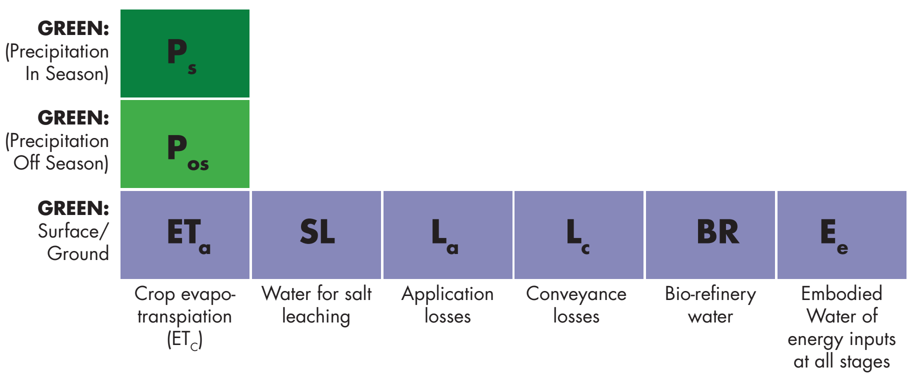
Another important aspect of the sustainability implications of different fuel/vehicle pathways has to do with their impacts on water resources. Production of fossil fuels, biofuels, electricity, and hydrogen all require consumption and/or withdrawal of freshwater to some extent.[9] However, determining the water footprints of different fuels is a complex topic, complicated by regional and seasonal variations in water availability or scarcity. A direct comparison between fuel pathways cannot be made simply by comparing totals or averages but must be examined at the local and regional level, considering water availability, water quality, and impacts on ecosystem health.

Water impacts of biofuel production

Mishra and Yeh assessed the water requirements of producing ethanol from corn grain and crop residue. They explicitly tracked volumes of water use by different categories throughout the life cycle, including evapotranspiration, application and conveyance losses, biorefinery uses, and water use of energy inputs. They also considered avoided water use due to co-products, which estimates the amount of water that would have been consumed without the production of co-products.

The two categories of water use the researchers examined were (1) consumption of blue water (BW, meaning surface or ground water) and green water (GW, meaning precipitation and soil moisture), and (2) withdrawal of blue water. Consumption is the use of freshwater that is not returned to the watershed but instead is lost as a result of evaporation, evapotranspiration, incorporation into the product, discharge to the sea, or percolation into a salt sink. Withdrawal is the removal of water from a surface water body or aquifer to be used both consumptively and nonconsumptively. BW used nonconsumptively is released back to the environment with or without change in quality, through recycling to water bodies, seepage, and runoff, and is available for alternative uses though these may be in different watersheds or at different times. Unlike BW, use of GW is considered only in a consumptive sense. Water usage is estimated in the form of liters per vehicle kilometer traveled (L/VKT) and hence referred to as water intensity.

LIFE-CYCLE WATER REQUIREMENTS OF BIOFUEL PRODUCTION



Mishra and Yeh tracked the water required to make ethanol from corn grain and crop residue, including crop evapotranspiration (ETc) or irrigation (ETa), process and cooling water consumed during ethanol conversion (BR, which is included in most water footprint studies), as well as water for uses that haven't been considered by other researchers. These include water for salt leaching (SL), application losses due to irrigation system inefficiencies (La), losses during conveyance of irrigation water (Lc), and water requirements of fuels (Ee)—diesel, electricity, natural gas, and coal—used during corn cultivation, storage, and distribution, and during ethanol production.

Mishra and Yeh focused on ethanol from corn grown in California (CA) and in the U.S. Corn Belt—Illinois (IL), Indiana (IN), Iowa (IA), Kansas (KS), and Nebraska (NE). These states together accounted for more than 50 percent of the corn produced in the United States in 2009 and are likely to witness significant increases in corn cultivation and production of ethanol from both grain and agricultural residue as a result of aggressive targets set forth in the federal renewable fuel standard and the low-carbon fuel standard. For IL, IN, and IA, only rain-fed corn was considered, which accounted for more than 97 percent of the corn produced in those states in 2009. For NE and KS, water requirements of ethanol from rain-fed and irrigation corn were considered separately. All corn grown in California is irrigated.

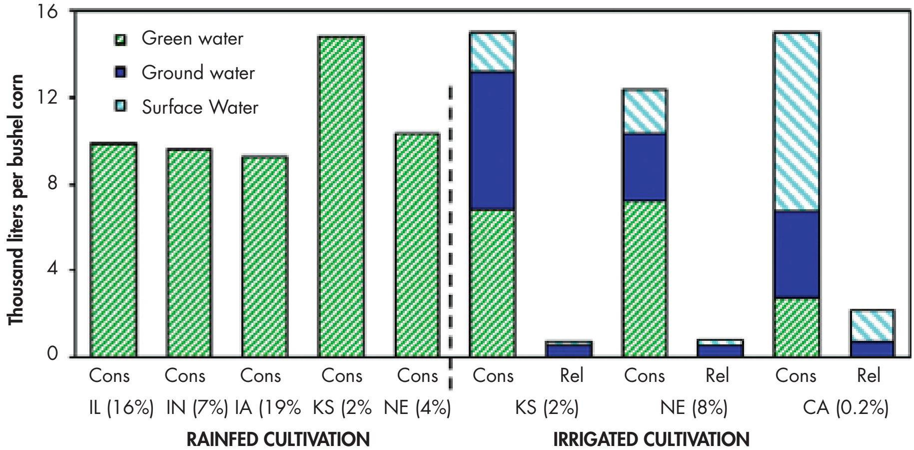
The researchers found that the GW consumption intensities of rain-fed corn in IL, IN, and IA are similar. The slight differences are entirely due to differences in yields, ET_c requirements, and supply constraints in the form of precipitation and available soil moisture. The team also found that irrigated corn yields are 50 to 60 percent higher than rain-fed yields in KS and NE, resulting in lower GW consumption intensity for irrigated corn in KS and NE, though the total GW and BW consumption intensities are roughly the same. In KS, water was applied at a rate of 40 centimeters (1.6 million liters per acre) for corn irrigation, which is 60 percent higher than in NE.

Though none of the previous studies considered nonconsumptive water withdrawal since the water is released back to the environment through recycling to water bodies, seepage, and runoff, ignoring such use fails to recognize that significant water withdrawals from surface water bodies may exert localized and/or seasonal impacts on the ecosystem. For regions dependent upon groundwater, extraction of groundwater beyond recharge rates could lead to aquifer depletion. Mishra and Yeh found that volumes of water returned (nonconsumptive water) in the form of seepage and deep water percolation account for 8 to 15 percent of total irrigation water withdrawn, which is attributable to the inefficiencies of furrow irrigation in CA and to the conveyance system (unlined irrigation canals) in NE. Most worrisome is that groundwater is the primary source of BW in both KS and NE, where it constitutes 60 to 80 percent of total water withdrawn. Evidence suggests that increased water use for corn is accelerating water-level declines in the Mississippi River Valley alluvial aquifer at an alarming rate.[10]

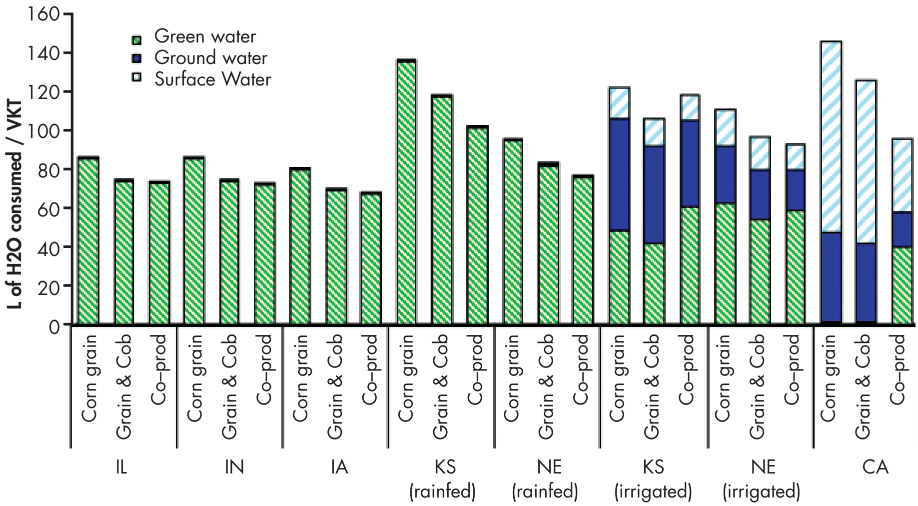
Since the production of solid, liquid, and gaseous biofuels may in itself generate co-products that displace other products requiring water for their supply, Mishra and Yeh contend that recognizing water requirements displaced by co-products significantly expands the system boundary of water use analysis and considers, albeit partially, the indirect water use associated with bioenergy expansion. In the United States, 88 percent of corn grain conversion to ethanol occurs through biochemical conversion using dry mill technology. A by-product of this process is distillers' grain soluble (DGS), which is used as an animal feed and can substitute for other animal feeds—namely corn grain, soybean meal (SBM), and urea. SBM in turn displaces raw soybeans. Production of DGS thus precludes the need to produce such other animal feed, so corn ethanol should be credited for water saved from not producing them. Similarly, electricity demands during production of cellulosic ethanol from cob are met internally through combustion of the lignin component of the cob, and the surplus—around 220 kWh/dry metric ton of cob—is exported to the grid. Surplus electricity is assumed to displace grid electricity, which has an average water intensity of 2.46 liters/kWh. However, very few cellulosic conversion technologies are currently operating commercially and data on ethanol yield and water consumption are uncertain.

Overall, ethanol from irrigated corn consumes 50–146 L/VKT of BW and 1–60 L/VKT of GW (90–211 L/VKT and 48–124 L/VKT of BW and GW respectively without co-product credits). For ethanol from rain-fed crops, the corresponding numbers are 0.6 L/VKT BW and 70–137 L/VKT GW (0.6 L/VKT and 140–255 L/VKT without co-product credits). Ethanol from cob consumes very little BW: 0.85 L/VKT after co-product credits. Harvesting and converting the cob to ethanol reduces both the BW and GW intensity by 13 percent.

LIFE-CYCLE WATER INTENSITY OF CORN, AND OF ETHANOL FROM CORN GRAIN AND RESIDUES



The volume of water required for corn cultivation—consumptive (Cons) and nonconsumptive (released water, Rel) use—is shown here. The values in parentheses are the share of corn produced in 2009. Irrigated corn grown in KS and NE has a 50- to 60-percent higher yield than non-irrigated corn but also requires more use of ground and surface water. The U.S. Geological Survey has found that increased water use for corn is accelerating water-level declines in the Mississippi River Valley alluvial aquifer.



The water consumption intensity of ethanol from corn grain versus grain and crop residue, and the avoided/displaced water use credits assigned to co-products, is shown here. These results suggest that harvesting and converting the cob to ethanol reduces both the BW and GW intensity by 13 percent. Cellulosic ethanol from cob only (not shown in the figure) has a BW consumption intensity of 0.85 L/VKT and zero GW intensity, which is entirely contributed from biorefinery water use. On average, co-product credits are around 5 percent and 45 percent of total BW used to produce ethanol from rain-fed and irrigated corn, respectively; and around 50 percent of GW in both cases. The results reflect the lower yields and hence higher water intensity of soybeans—for example, statewide average applied water for soybean cultivation was around three-quarters that of corn in 2008, but average dry matter yield was less than 40 percent.

Mishra and Yeh also estimated the water consumption of large-scale biofuel production at the state level and found that without accounting for co-product credits, 13 to 15 percent of irrigation water is used to produce the corn required for ethanol in the states of KS and NE, and 7 to 8 percent after credits. In IL, IN, and IA, where corn is largely rain-fed, biorefinery water consumption is less than 0.5 percent of overall BW use.

The researchers argue that the marginal effects of water requirements will be higher given the renewable fuel standards, which have led to higher corn prices as a result of ambitious production mandates. Higher corn prices could lead to expansion of corn production to marginal lands with lower yield potentials. It could also result in intensification of corn cultivation on existing lands, which could lower future yields. Since water intensity is negatively correlated with yield, such expansion and intensification will increase the water intensity of ethanol. Further, corn expansion is occurring disproportionately on land that requires irrigation, which according to these researchers' results has higher average total water due to seepage, application and conveyance losses (GW+BW) and irrigation water consumptive intensity, as well as high nonconsumptive water requirements due to seepage, application and conveyance losses.

Water use associated with other fuel pathways

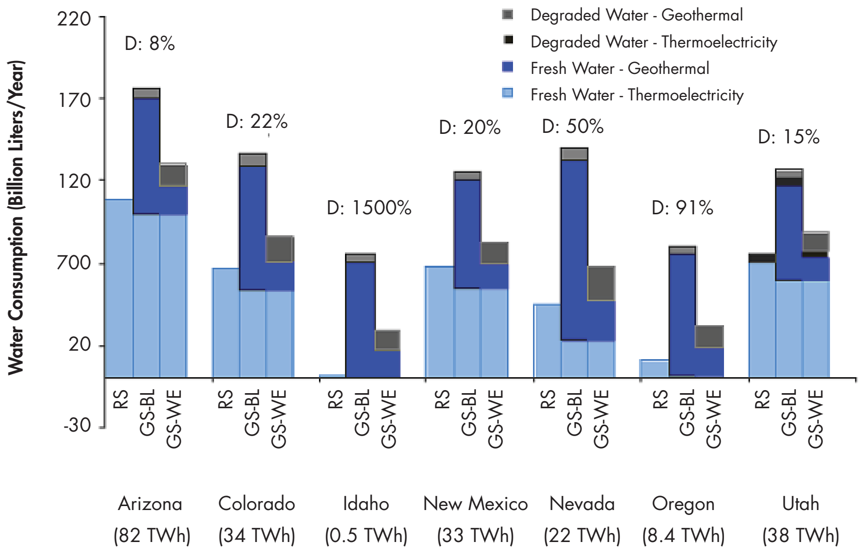
Water is also used in fossil fuel production. The BW consumption intensity of gasoline from conventional crude oil and Canadian oil sands is 0.41–0.78 L/VKT and 0.29–0.62 L/VKT, respectively.[11] Oil recovery using technology such as water flooding, enhanced oil recovery (EOR) via steam injection, and oil sands in-situ production is the major water consumption step in the petroleum gasoline life cycle. A recent report from the U.S. Government Accountability Office suggests the water intensity of gasoline from shale oil from large deposits found in Colorado, Utah, and Wyoming could be in the range of 0.29–1.01 L/VKT.[12]

Electricity production also withdraws a large amount of water, but the amount of water withdrawn and the impacts on water resources vary by region. The average water withdrawal intensity of thermal electric plants in a region typically correlates with the amount of water resources available within the region. Therefore, an important consideration for assessing the water resource impacts of fuels is the relative water intensity compared to the regional water shortage level. In addition, technology choices, water management, and technological change also explain variation in water use. The national average freshwater withdrawal per unit of electrical energy has decreased more than 35 percent since 1985 despite an increase in the total electricity produced, resulting in the total thermal electric freshwater withdrawal remaining constant over the same period.

Similar to the work on biofuels, Mishra, Glassley and Yeh[13] estimated the fresh and degraded water requirements of geothermal electricity. The research found that geothermal electricity is, in general, less water-efficient than other forms of electricity such as coal- and gas-fired power plants and renewables like solar thermal (i.e., water requirements of electricity from geothermal resources are substantially higher than those of both thermoelectricity and solar thermal electricity for the same amount of electricity generated). Mishra, Glassley and Yeh also conducted a scenario analysis to measure the potential impact of potential scaling up geothermal electricity on water demand in various western states with rich geothermal resources but stressed water resources. Electricity from enhanced geothermal systems (EGS) could displace 8–100% of thermoelectricity generated in most western states.[14] Such displacement would increase stress on water resources if re-circulating evaporative cooling, the dominant cooling system in the thermoelectric sector, is adopted. Adoption of dry cooling, which accounts for 78% of geothermal capacity today, will limit changes in state-wide freshwater abstraction, but increase degraded water requirements.

The research by Mishra, Glassley and Yeh identified the need for R&D to develop advanced geothermal energy conversion and cooling technologies that reduce water use without imposing energy and consequent financial penalties. Further, their results highlighted the need for policies to incentivize the development of higher enthalpy resources, and support identification of non-traditional degraded water sources and optimized siting of geothermal plants.

WATER IMPACTS OF DISPLACING THERMOELECTRIC WITH GEOTHERMAL ENERGY IN FOUR WESTERN STATES



The figure above estimates the impact of displacement of thermoelectricity by EGS electricity on consumptive water requirements. The percentage of thermoelectricity produced in reference scenario (RS) and displaced by electricity from Enhanced Geothermal resources in Geothermal Scenario (GS) is represented by "D". Two geothermal sub-scenarios are envisaged—the baseline (GS-BL) and water efficient (GS-WE) scenarios. In the GS-BL scenario, where evaporative re-circulating cooling dominates, statewide water requirements increase substantially. In the GS-WE scenario, where dry cooling is used in 78% of geothermal electricity, as is the scenario today in the U.S., water requirements increase by a smaller magnitude.

Overall comparisons

Direct comparison of the water demands of biofuels and fossil fuels is much more complicated than simply comparing a commonly used, yet oftentimes erroneous due to its simplicity, water footprint indicator. The BW consumption of biofuels from rain-fed crops and residue is lower than that of gasoline, but it is orders of magnitude higher if the biofuels are from irrigated crops. Ethanol from corn grain has a high groundwater requirement, and groundwater use impacts terrestrial ecosystems and BW availability. Though the water intensity of fossil fuels is on average low compared with biofuels, it has been widely reported that oil sands production and potential shale oil development could result in substantial streamwater withdrawals and significant alteration of water flows during critical low river flow periods; groundwater depletion and contamination; and wastewater discharges.[16] A detailed comparison of biofuel versus fossil fuel water use should carefully examine the impacts of water use on changes in water availability and quality and other ecosystem health effects at the local level and/or accounting for season variability, though such comparison is often missing in the literature and also unfortunately beyond the scope of this analysis.

Mishra and Yeh caution that their assessment necessarily employs spatial and temporal aggregation by summing across types of water consumption (BW and GW consumption and avoided water credits) in locations where the relative importance of water-related aspects may differ; thus, some results may carry no clear indication of potential social and/or environmental harm or trade-offs. Similarly, temporal aggregation of water use estimates ignores the interseasonal variability of water use and water scarcity and can therefore yield erroneous conclusions concerning seasonal water use competition. Recent literature on freshwater life-cycle analysis has developed regionally differentiated characterization factors that measure water scarcity at a watershed level and also account for temporal variability in water availability. For example, in future studies volumetric estimates of green and blue water can be converted to characterization factors, providing a "stress-weighted" or "ecosystem-equivalent" water footprint estimate that can be compared across regions. Such work is still ongoing.[17]

Anticipating Material Use in New Vehicle Technologies

In a sustainable transportation system, the key new technologies will be electric motors and controllers, batteries, and fuel cells. An important question is whether any of these technologies use materials that are either scarce or else concentrated in a few countries and hence subject to price and supply manipulation, in which case the need for such materials might become a barrier to development. Here we focus on rare-earth elements (REEs) for electric motors, lithium for lithium-ion batteries, and platinum for fuel cells.[18]

Neodymium for electric motors

Some permanent-magnet alternating-current motors can use significant amounts of REEs. For example, the motor in the Toyota Prius uses 1 kg of neodymium (Nd) or 16-kg/MW (assuming that the Prius has a 60-kW motor).[19] In a worldwide fleet of EVs with permanent-magnet motors, the total demand for Nd might be large enough to be of concern, especially because permanent-magnet motors with Nd are also used in generators for wind-power turbines. A highly electrified world in which 50 percent of global electricity was provided by wind turbines and two-thirds of light-duty vehicles had electric motors could require up to 200,000 metric tons of Nd oxide per year. This rate of consumption would exhaust known global Nd-oxide reserves in less than one hundred years and would exhaust the more speculative potential resource base in perhaps a few hundred years. Therefore, it seems likely that a rapid global expansion of wind power and electric vehicles eventually will require generators and motors that do not use Nd or other REEs. However, this is not likely to be a serious constraint, because there are a number of alternatives to Nd for use in motors and generators.

Lithium for batteries

Roughly half of the world's identified lithium resources are in Bolivia and Chile. However, Bolivia does not yet have any economically recoverable reserves or lithium production infrastructure, and to date has not produced any lithium. A little more than half of the world's known economically recoverable reserves are in Chile, which is also the world's leading producer. Both Bolivia and Chile recognize the importance of lithium to battery and carmakers, and are hoping to extract as much value from it as possible. This concentration of lithium in a few countries, combined with rapidly growing demand, could cause increases in the price of lithium. In 2010, lithium carbonate (Li_2CO_3) sold for \$6–7/kg, and lithium hydroxide (LiOH) sold for about \$10/kg, prices which correspond to about \$35/kg-Li. Given that lithium is 1–2 percent of the mass of lithium-ion batteries, a battery in an electric vehicle with a relatively long range (about 100 miles) might contain on the order of 10 kg of lithium. At 2010 prices this amount of lithium would contribute \$350 to the manufacturing cost of a vehicle battery, but if lithium prices were to double or triple, the lithium raw material cost could approach \$1,000. This could have a significant impact on the cost of an electric vehicle.

If one considers an even larger electric vehicle share of a growing future world car market and includes other demands for lithium, it is likely that the current lithium reserve base will be exhausted in less than twenty years in the absence of recycling. As demand grows the price will rise, and this will spur the hunt for other sources of lithium, most likely from recycling. The economics of recycling depend in part on the extent to which batteries are made with recyclability in mind. Ultimately the issue of how the supply of lithium affects the viability of lithium-ion-battery electric vehicles boils down to the price of lithium with sustainable recycling.

Platinum for fuel cells

The production of 20 million 50-kW fuel cell vehicles annually might require on the order of 250,000 kg of platinum (Pt)—more than the total current world annual production. How long this output can be sustained, and at what platinum prices, depends on at least three factors: (1) the technological, economic, and institutional ability of the major supply countries to respond to changes in demand; (2) the ratio of recoverable reserves to total production, and (3) the cost of recycling as a function of quantity recycled.

The effect of recycling on platinum price depends on the extent of recycling. It seems likely that a 90-percent-plus recycling rate will keep platinum prices significantly lower than will a 50-percent recycling rate. We cannot predict when and to what extent a successful recycling system will be developed. Nevertheless, we believe that enough platinum will be recycled to supply a large fuel-cell vehicle (FCV) market and moderate increases in the price of platinum, until new, less costly, more abundant catalysts or fuel-cell technologies are found. Indeed, catalysts based on inexpensive, abundant materials may be available relatively soon; research on iron-based catalysts suggests that a worldwide FCV market will not have to rely on precious-metal catalysts indefinitely.[20]

Preliminary work by Sun et al. (2010) supports this conclusion that platinum recycling will moderate the cost of platinum for FCVs.[21] They developed an integrated model of FCV production, platinum loading per FCV (a function of FCV production), platinum demand (a function of FCV production, platinum loading, and other factors), and platinum prices (a function of platinum demand and recycling). Based on this model, they found that in a scenario in which FCV production was increased to 40 percent of new light-duty vehicle output globally in the year 2050, the average platinum cost per FCV was \$500, or about 13 percent of the cost of the fuel-cell system.

Summary and Conclusions

- This chapter has explored sustainability issues associated with land, water, and materials impacts of production along alternative fuel pathways compared with petroleum-based gasoline and diesel. The studies discussed here are representative, but this discussion is by no means comprehensive. It is also important to note that much work remains to be done on understanding and measuring these impacts.

- When biofuels are produced on carbon-rich lands such as forest or tropical peatlands, the resulting GHG emissions may take decades of biofuel production to sequester back. Because biofuels require orders of magnitude more land than do petroleum fuels for the same amount of energy produced, land-use GHG emissions per unit of energy output can be significantly higher than for oil. This is aside from the issue of indirect land-use impacts of biofuels, which are considered in Chapter 12.

- GHG emissions from land-use disturbance caused by fossil fuel exploration and extraction can be significant. In heavily mined areas after oil sands production has been completed, efforts should be focused on post-mining reclamation such as the restoration of habitat to reduce land-related CO_2 emissions, recover ecological landscapes, sustain high biodiversity, and maintain hydrologic cycles and forest ecosystems.

- The sustainability impacts of fuel production on water resources need to be compared at the local and regional levels. Concerns about local impacts on water availability, water quality, and ecosystem health should be carefully evaluated. The relative importance of water aspects compared to other aspects of the shift to a new transportation energy system—such as effects on GHG emissions, soil quality, biodiversity, and economic sustainability—must be weighed.

- The research on life-cycle material use by new vehicle technologies suggests that it is unlikely that material use will impose serious constraints on technology development in the long term. However, short-term price volatility and sustainability impacts due to extraction activities need to be considered and mitigated whenever appropriate.

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