Energy Research and Development Division
FINAL PROJECT REPORT

WATER USE IMPLICATIONS OF CALIFORNIA'S FUTURE TRANSPORTATION FUELS

Appendices A - G

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TABLE OF CONTENTS

TABLES

FIGURES

Appendix A. Water Use LCA – Methodological Debates & Frontiers

In an era of continuing population growth and rapid economic activity, together with anthropogenic climate change, increasing variability and scarcity of water supplies has driven renewed attention to quantifying use and characterizing the impacts of extracting and altering this vital resource. However, despite on-going efforts to reach consensus on methods for water use in lifecycle assessment (WULCA), no single methodology has been accepted as adequate to inventory and assess the impacts of water use. The difficulty in estimating water volumes can be attributed in part on incomplete data and uncertain measurements. But another hurdle is methodological, and may reflect inaccurate conceptualization of a particular volume of water according to its *origin*, *fate*, *transformation*, and *functionality*. Difficulties in determining water quality and quantity in ecological, industrial, and agricultural settings arise when tracing complex stocks and flows that are highly time- and space-dependent. Similarly, difficulties persist in conceptualizing non-overlapping, consistent, and causal linkages tying water quality degradation to its human health impacts.

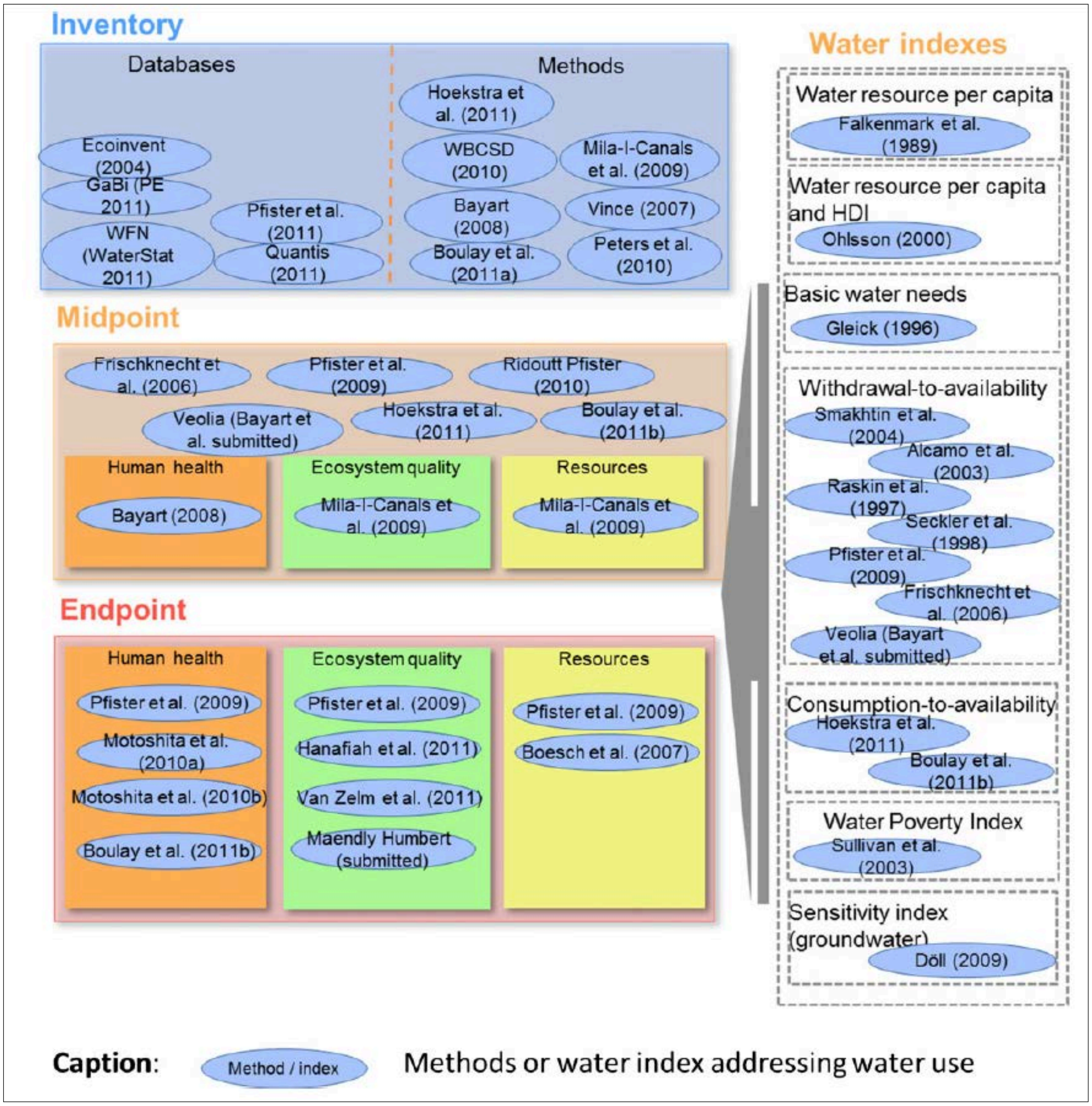
WULCA is plagued by constraints on accurate data availability. Restrictions on data access and uneven reporting, and in particular poor characterization of global groundwater stocks, both within the U.S. (Iwanaga, El Sawah et al. 2013, Schreiber 2013) and particularly at a global level (Fan, Li et al. 2013, Gleeson and Wada 2013, Jin and Feng 2013). This makes estimating the volumes and quality impacts of fresh water use particularly difficult. The proposed midpoint level methodologies to characterize water quantity and quality impacts on human health endpoints are subject to tradeoffs between data availability and output resolution. Therefore, recommendations are made to improve watershed level reporting and characterization methods in effort to enhance the use of Lifecycle Assessment analyses in water characterization of energy supply chain operations.

A.1 Overview of Existing WULCA Methods

The following 'meta-review' draws heavily on and seeks to build upon two recent reviews of WULCA methods (Berger and Finkbeiner 2012, Kounina, Margni et al. 2012), as well as from other recent key methodological developments in WULCA, such as the latest Water Footprint Manual (Hoekstra, Chapagain et al. 2009) and (Boulay, Bouchard et al. 2011). The review papers classify various water quantity and quality impact assessment approaches at the midpoint and endpoint level in WULCA. They describe and contrast databases and indexes developed over the last decade. To illustrate the sheer number of methods proposed, the classification of Kounina, Margni et al. (2012) is reproduced in Figure 1.

General observations of current methods for estimating water impacts include the often inadequate or inconsistent delineate a particular volume of water's source, fate, use, and alterations in quality/functionality. Difficulties in tracking water quality and quantity can be attributed to the various functions and services performed in ecological, industrial, and agricultural settings; to complexity in tracing stocks and flows; and to the highly time- and space-dependent availability (scarcity) of the resource. Further, inclusion and characterization of non-local water use impacts is inconsistent and WULCA often ignores the water used in background processes (e.g. extraction of energy feedstocks for electricity generation). Following a discussion of the foundations of WULCA, methods amendments are proposed to address observed shortcomings relevant to water use impacts of energy supply chains.

Figure 1. Proposed Methods in Water Use Lifecycle Analysis (WU-LCA)



Reproduced from Kounina, Margni et al. (2012).

Water Quantity Methods

State-of-the-practice review

An established set of methods has emerged for basic LCA inventory of freshwater use over the past decade. Freshwater is defined as water with a salinity concentration of less than 1000 mg/L. The most basic classification is between blue- and green- water, and consumption and withdrawal. By definition consumptive water use entails removal, via product incorporation, evaporation, plant transpiration, or transfers (e.g. via aqueducts or pipelines) of a volume of water from the region (ideally a watershed). Thus, green water consumption is water sourced from soil (ultimately via rainfall) and evapo(transpi)red by plants. Blue water consumption includes that portion of surface- or groundwater sourced for crop cultivation via irrigation that is evapo(transpi)red, while the portion commonly considered 'lost' to the soil is considered blue water withdrawal. Green- and blue- water withdrawals/consumption accounting may follow either source-based (Hoekstra, Chapagain et al. 2009, Berger and Finkbeiner 2012) or use-based methods (IWMI 2007). The distinction is relevant for agriculture, and the source-based approach is preferable, as it recognizes that the functional limitations of water stored as soil moisture (i.e. that it is only available to plants), and hence provides useful information to water users and managers.

In the context of electricity generation, blue water use is further differentiated into in-stream versus out-of-stream withdrawals. Consumption, in this case, is evaporation, while withdrawal may incur quality degradation (increased salinity of blowdown water in the case of recirculating cooling, or thermal pollution in the case of once-through cooling, etc.), which is ideally included in a proper WULCA of electricity generation.

Water withdrawal may further incur changes to water quality, and within some methodological frameworks this is considered 'freshwater degradative use.' In an effort to report all inventories as quantities, quality impacts are sometimes reported volumetrically as gray water, which is the volume of water that would be needed to dilute degraded water withdrawals to achieve regionally regulated quality standards (e.g. for drinking, agricultural, or industrial use). The gray water designation has fallen out of favor, and many have pointed out its several drawbacks, such as the dependence on variable local standards, the arbitrary and inconsistent conversion of qualitative information to quantities (Kounina, Margni et al. 2012), and the overlapping terminology with the more widespread use of 'greywater' as nutrient-rich recycled water, not containing fecal matter, that is suitable for (particularly agricultural) reuse (Berger and Finkbeiner 2012).

Water used in background processes should be accounted for in a full lifecycle analysis. In WULCAs tracking production and global trade, background water use has been termed "virtual water." Correct quantification should include both water incorporated in actual materials and water used in the background process itself. Recognizing that even if virtual water use is accounted for, it carries little meaningful information without explicit data on source and flows, Hoekstra and Hung (2002) quantified virtual water flows including spatial and temporal data. While their effort was limited by data availability and so reported as nationally aggregated estimates, it recognized the necessity of such information. In the interim, watershed resolution estimates have been developed (Hoekstra, Chapagain et al. 2009, Boulay, Bulle et al. 2011).

Database Review

A handful of WULCA databases exist, but are not consistent with one another and each has certain limitations. Pfister, Bayer et al. (2011) and the Water Footprint Network's (Hoekstra, Chapagain et al. 2009) databases distinguish between blue- and green-water, but make no further differentiation among bluewater sources (i.e. surface water from lake-, river- or stream-source; or groundwater renewable or fossil aquifers). Ecoinvent (2013) and GaBi (2013) both additionally distinguish among blue water source bodies to the degree data availability permits. However, Ecoinvent measures freshwater inputs but not wastewater outputs, and so does not permit differentiation of consumption vs. withdrawal. Further, as noted by Berger and Finkbeiner (2012), GaBi (PE_International 2013) differentiates between withdrawal and consumption, but does not include background process ('virtual') water use from upstream operations like mining, processing, and concentration/refining of energy and material feedstocks. The World Business Council for Sustainable Development's Global Water Tool (WBCSD 2013), provides a database tailored to provide information to industries, and includes spatial information on major surface water stocks, population densities, withdrawal estimates (including specialized tools for global oil and gas and electricity generation), and derived estimates of scarcity. Both WBCSD (2013) and (Boulay, Bulle et al. 2011) provide input-output inventories of water flow at high spatial resolution, and Boulay's method currently provides the most highly disaggregated publicly available database to estimate water among eleven elementary flow classifications and quality based on 137 parameters (Kounina, Margni et al. 2012).

Methods Review

Divergence still exists among methods for WULCA inventory and even more for characterizing the impacts of freshwater use. Despite some preliminary recommendations by the United Nations Environmental Program / Society of Environmental Toxicology and Chemistry (Rosenbaum, Bachmann et al. 2008) Life Cycle Initiative's Water Use LCA working group, many issues are still unsettled (Kounina, Margni et al. 2012), and even some of those that are recommended communicate information of dubious value (Berger and Finkbeiner 2012).

Kounina, Margni et al. (2012) leverage their positions as members of the UNEP/SETAC Life Cycle Initiative's Water Use LCA (WULCA) working group to undertake a "detailed and systematic" analytical review of freshwater LCA methods. Their criteria for recommending certain methods include "scientific robustness, transparency and reproducibility, applicability, the level of documentation, and stakeholder acceptance." They stress that their recommendations concerning LCA and LCIA methods represent a scientific consensus, and thus represent an improvement over previous methods reviews (Berger and Finkbeiner 2010).

Based on the deliberations of a large panel of WULCA experts, (Kounina, Margni et al. 2012) recommend that databases be standardized and updated to report water flows differentiated to the extent allowed by input data availability, across the following metrics:

(1) *Type*: either blue water in the form of either surface (split by river, lake, or sea) or groundwater (shallow vs. deep renewable, and, fossil) stocks; or green water (sourced by soil moisture);

(2) *Quality*: organic and inorganic compound concentrations, temperatures, general parameters like BOD and COD as well as microbe presence/absence or concentrations, and any other relevant and readily available water quality information, following the listings of Boulay, Bouchard et al. (2011) as a general guideline;

(3) *Technical water flows*, such as cooling, irrigation, or produced water;

(4) *Consumption* vs. *Withdrawal*: with the former including evaporative estimates from surface water bodies.

Kounina, Margni et al. (2012) further suggest regionalization of databases for LCA, though without specifying the recommended resolution. This ambiguity is perhaps intentional, and likely intended to indicate that databases to the highest resolution permitted by data availability. While this reflects a realism in the face of constrained resources, given the primacy of water resources to economic development, it seems that the ideal resolution is at the watershed level, but that connections among watersheds should be modeled spatially in any impact assessment to afford maximum information and realism to analysts, water managers, and decision makers.

To characterize impacts of freshwater withdrawal/consumption, Kounina, Margni et al. (2012) favor midpoint level impact assessment methods relying upon a scarcity index as measured in terms of water consumed or withdrawn as a fraction of total water volumes available within a watershed, but note that more refined midpoint metrics distinguish among water types and quality, following the methods developed by Boulay, Bouchard et al. (2011). They recommend that scarcity indices should include water storage capacity whenever such data are available.

Finally, Kounina, Margni et al. (2012) review proposed endpoint assessment methods in each of the three areas of protection (human health, resources, and ecosystem quality), but note that no comprehensive and causally viable methods for characterizing impacts have been developed for any of the areas. Given data limitations, uncertainties, and the current incomplete methodological state of WULCA, they recommend characterization based on midpoint assessment.

Berger and Finkbeiner (2012) organize their contributions to the debate on water use LCA methods by dividing methods into two categories: (1) volumetric studies that simply report categories of water use and (2) impact-oriented assessments that aim to report the widely varying impacts of water use, as determined by scarcity, quality, accessibility, and infrastructure and socio-economic considerations (such as scaling impacts using the Human Development Index). These authors level a number of cogent critiques at methodological flaws in the current WU-LCA framework. Most relevant to characterization of water consumption in fossil fuel extraction are their objections to the two most common water scarcity indices: the "withdrawal-to-availability (WTA) ratio" and the "consumption-to-availability (CTA) ratio." A simplified formula for WTA / CTA is:

$$\mathrm{WTA}_{\mathrm{i,j}} = \frac{\text{water withdrawal rate}_{\mathrm{i,j}}}{\text{total availabile stock}_{\mathrm{i,j}}} \qquad \mathrm{CTA}_{\mathrm{i,j}} = \frac{\text{water consumption rate}_{\mathrm{i,j}}}{\text{total availabile stock}_{\mathrm{i,j}}}$$

where subscripts *i* and *j* denote source/type and quality/functionality classifications, respectively, and the total available stocks are a function of renewability rates for each source. The CTA was proposed after the WTA as a metric that more accurately reflected scarcity. However, both indices share a number of shortcomings. Neither conveys information on marginal impacts of a new withdrawal/consumption process accurately – rather, only the proportion of water used is communicated. To convey absolute scarcity (i.e. a small magnitude of the denominator), Berger and Finkbeiner (2012) recommend reporting minimum threshold value such as rainfall less than 200 mm/year, or precipitation < 5% of PET. In addition, it may be advisable to report the percentile of stock availability by water type/source in terms of global distribution. This would convey the absolute biophysical water availability, by stock, for a given region, usefully independent of socio-economic factors such as levels of industrial or agricultural activity and population density. Finally, neither method accounts for temporary resilience and recharge management measures such as groundwater banking. While such information would be valuable, considerations of storage capacity such as those done by Pfister, Koehler et al. (2009) are also limited by lack of data on the volumes and recharge rates of groundwater stocks.

Berger and Finkbeiner (2012) further challenge the validity of naively reporting changes ('deltas') of hydrological balances due to land use change. Clearly increased soil water recharge does not necessarily translate to benefits over increased availability, as is implied by withdrawal/consumption to availability ratios – the authors cite the examples of flooding and waterlogging. Instead LCIA characterization must be contextualized, by taking into account water use patterns, infrastructure, management, and typical seasonal volumes and flows.

Appendix B. Further Details on Water Use of Hydraulic Fracturing

B.1 Regulation and Classification of Oil and Gas Injection Wells

Large volumes of salt water (brine) are produced in oil and gas production. Often this water contains a mix chemicals used to stimulate fossil fuel production, and it may also contain trace concentrations of heavy metals and/or radioactive elements in the geological formation. It is thus in the public interest for maintaining clean groundwater supplies (e.g. for drinking water) to reinject produced water deep underground. Produced fluids must be injected into Class II injection wells. The EPA regulates injection of produced water under its Underground Injection Control (UIC) program, which mandates permitting and monitoring of Class II wells. The EPA estimates that there were around 144,000 Class II wells operating in the U.S. in 2012, injecting an average of more than 2 billion gallons of salt water per day (EPA 2012). Three categories of Class II well exist:

Enhanced recovery wells (secondary or tertiary recovery wells) inject fresh and/or salt water, steam, polymers, or CO_2 in cases where it is technically and economically feasible to reuse and/or recycle some proportion of produced water to reduce viscosity and/or increase pressure in the oil/gas well, and induce oil/gas production. Typically, a single enhanced recovery well is surrounded by production wells. The EPA estimates that enhanced recovery wells make up about 80% of injection wells in the U.S.

Disposal wells are used to dispose of the majority the salt water produced in oil/gas production. They account for most of the remaining 20% of the injection wells.

Hydrocarbon storage wells use natural geological formations, such as salt caverns, to store hydrocarbon reserves. Of the more than 100 liquid hydrocarbon storage wells across the country, most are part of the U.S. Strategic Petroleum Reserve.

B.2 The Domestic Shale Oil Boom

Between 2011-2012, the number of tapped U.S. shale oil plays jumped from a few hundred to more than 4,000 (Maugeri 2013). A resource that was virtually untapped in 2006 produced just short of 2.0 million barrels per day of crude oil by the end of 2013 (EIA 2013d). Total reserve estimates are huge – the Bakken Formation alone may hold more oil in place than Saudi Arabia – but rapid declines in oil produced by a given well lead many to believe that only a small fraction of these reserves is recoverable (Maugeri 2013). The U.S. EIA projects that the U.S. has the potential by as early as 2017 to become the world's top oil producer, and could reach virtually complete self-sufficiency (97%) in supplying net energy demand of oil (EIA 2013). According to the most recently published U.S. EIA estimates, roughly 13.7 billion barrels of oil may be locked within the Monterey Shale (EIA 2012a).

As with shale gas, production in shale oil wells peaks rapidly – within the first 30 days – and then declines by 40-50% by the end of the first year, and by another 30-40% by the end of the second year (Maugeri 2013). On the basis of estimates compiled by Maugeri (2013) and EIA (EIA 2013c) projections of shale oil production to 2030, the weighted-average recovery rate of shale oil across the "Big Three" oil plays (the Bakken, Eagle Ford, and Permian Basin) five years after well completion is estimated at around 14% of peak well production (assuming current technology and operations).

B.3 Recommended Methods for LCA Inventory and Impact Assessment

Background water use, including water use in drilling and proppant mining, should be included in a comprehensive LCA, as these may account for roughly one-quarter of water consumption in horizontal fracking in shale gas production (Nicot and Scanlon 2012). If operation-specific volumes can not be obtained, research may rely upon literature estimates for drilling water use (Nicot and Scanlon 2012), Goodwin et al. (Goodwin, Carlson et al. 2012) and proppant mining, as normalized by average proppant loading values (Nicot and Scanlon 2012). However, literature estimates for drilling WUI should be used with care, as rapid technological development has translated to rapid reductions in WUI for drilling over the past decade (Nicot and Scanlon 2012).

The scarcity of data on water quantities consumed renders impossible comparison of water use across various plays or a national assessment of the water use impacts of fracking. To assess the systemic impacts, including price-induced fuel switching from other water intensive and highly polluting fossil energy feedstocks, more state regulatory agencies need to require reporting of water volumes used for both hydraulic fracturing completions and refracking. Some states have implemented such regulations, which should be the minimum data collected by state regulators. Ideally, water sources and basic data on water quality should also be reported, and independent verification should be conducted by state environmental protection agencies. Of the three articles reviewed in the water quantity section, only Murray reported produced water ratios (Murray 2013). Due to concerns over potential water quality impacts of produced water during storage, transport, and treatment (Cooley, Donnelly et al. 2012), as well as evidence associating water reinjection with seismic events (Aldhous 2012, Ehrenberg 2012) operators should be required to report annual volumes of produced water treated and reinjected.

Water volumes used for hydraulic fracturing are too often reported only on a per well basis. For watershed level analysis, it is important to report on such a basis, and further to briefly characterize, and provide references to a more detailed description of, the attributes of the geological formation(s) being exploited (e.g. rock porosity and type e.g. shale/tight; estimations of the resource volume and extent), as well as of the well (e.g. depth and cubic meters per perforated meters), and most importantly the names and respective volumes abstracted from each surface/groundwater body. But unless these consumed water volumes are referenced to (1) actual, and (2) expected ultimate oil/gas recovery (EUR) values, no comparison can be made among the WUI of (horizontal) hydraulic fracturing and other fossil fuel production operations. Similarly, aggregated water use must reported at a county (or better, watershed) level, so that current or projected impacts on local water availability can be estimated, as in the analysis conducted by Nicot and Scanlon (Nicot and Scanlon 2012). Finally, impact indices such as WTA and CTA (discussed above) should be applied to the volumes, absolute water availability reported (e.g. in terms of global percentiles), and a discussion should attempt to contextualize the impacts of land use change incurred by installation of production platforms within the hydrological setting of the operation.

The primary limitation of the recommended water quantity and quality methodologies is that they are constrained by the lack of available data. In cases where data on water volumes is reported, information on the water source and quality is rarely provided. There is no independent verification of consumptive or produced water volumes. In cases where reporting is voluntary, it is likely that values are not representative of operations in a given area; rather, only the operators with the highest standards are likely to report voluntarily. Hydraulic fracturing specifically is regulated differently between states making consistent data collection difficult, and comparison between operations problematic. This study has taken steps to identify data inputs necessary for watershed level quantity and quality evaluation of water use. As the growth of the natural gas industry progresses, now is the time to set national standards on data collection to monitor water use and quality to ensure that water scarcity and human health impacts are not an insurmountable concern moving forward.

Preliminary comparison of the water use intensity (WUI) of horizontal hydraulic fracturing with conventional oil and gas and coal production, as well as mining of radioactive materials for nuclear power (Nicot and Scanlon 2012) as well as a methodologically flawed comparison to renewable energy sources (Goodwin, Carlson et al. 2012) have been conducted. These may be misleading, however, for the following two reasons. First, in terms of impact of fossil fuel production's impact on ecosystems, human health, and the local resource base, it is the watershed/county level impact counts, not intensity. This may be taken as an argument *for* conducting impact analyses – despite the incompleteness and subjectivity of the current state-of-the-practice. Secondly, to adequately compare among alternative energy supply chains, it is necessary to conduct a *systemic* analysis. A suggestive study in this vein is offered by Murray (Murray 2013), who cites Grubert, Beach et al. (2012) as showing that in Texas some portion of the projected increasing volumes of freshwater consumed for fracturing in oil/gas production is offset by switching from coal power plants to more energy and water efficient NGCC utilities. A comprehensive systemic analysis would examine the impacts of a range of energy supply scenarios using comprehensive LCA techniques applied to attributional LCA WUI values (i.e. a combination of hybrid LCA and scenario analysis). However, this later approach is wholly unfeasible given the current state of ignorance concerning the quantity and quality impacts of horizontal fracking. Further, scenarios would require treatment of the many uncertainties in how the industry will develop in the coming decade given the web of rapidly evolving regulatory, economic, and technological factors that will shape is evolution.

Another aspect of fracturing not analyzed in any of the articles reviewed is the practice of refracturing. Wells are not uncommonly refractured in the hopes of stimulating further production – though the success of the technique is not ensured. Unfortunately, the relative rates of failure and success in refracking are not known, and the WUI relative to original completion fracking has not been investigating on either a per well or net energy basis.

Without significantly stronger regulations or increased accessed and scientific scrutiny, it is likely that WULCA will have to continue to rely on imprecise and unsubstantiated volumes reported to evaluate the water impacts of horizontal hydraulic fracturing. Thus, a tradeoff that perpetually arises in WULCA between 'precision' and 'applicability' (Berger and Finkbeiner 2010), is all the more pronounced in this case. In addition to questions of data quality and availability, this tradeoff will define the evolution of WULCA methods and limit the spatial resolution at which studies can be performed.

The methods recommended above for estimating volumes of water consumed in well completion and produced water apply broadly across all fossil fuel production processes. However, in the case of surface heavy oil production (e.g. tar sands) and coal mining, process specific water quality impacts may be incurred (e.g. leaking from holding ponds, runoff from drilling operations, or spills in transport to treatment plants).

B.4 Review of Published Studies on Hydraulic Fracturing of Oil and Gas

Table 1 provides an overview of the published academic literature tracking the water volumes used in domestic horizontal hydraulic fracturing ("fracking") operations. Note that all of these studies focus on shale/tight *gas*, some of which is coproduced with oil. Thus, to date, no study has been published examining the water use incurred by fracking of shale/tight *oil* resources.

Table 1. Water Use (by Activity) for Conventional Drilling and Hydraulic Fracturing.

Well type	m^3 H_20 per well (median)
	nd not on EUR ratios over the well lifetimes.

Appendix C. Oil Production in California

C.1 Energy Allocation between Oil and Natural Gas

The estimated *Net* and *Total* water use intensity values reported here are allocated to oil production alone. However, in the case of coproduced oil and natural gas, water is also allocated first to oil alone, and then according to the combined energy content of the ***produced and marketed*** energy carriers (i.e. oil *and* gas), as explained in this section. Estimated WUI values based on this energy content allocation method are available on request.

In the case of tertiary production, any produced natural gas is first used on site for steam production, after which unused gas is sold on the market (Brandt 2011). In 2012, the median value of natural gas to oil produced was 0.013 MJ gas/MJ oil, and the production-weighted average was 0.74 MJ natural gas/MJ oil. To properly attribute net water use to oil and gas, water use is allocated to oil only (upper bound for WUI) and oil & gas production (lower bound for WUI) by extraction technology.

Total and Net Water Intensity of Secondary Oil Production

Water allocation for oil is conducted differently for secondary and tertiary production. For secondary extraction, or water flooding, net water use is divided by total oil produced (*Method 1*) or water is allocated in proportion to the energy content of oil and gas produced (*Method 2*). All the values derived and reported in the body of the report are based on *Method 1*, though values for *Method 2* can be obtained upon request.

Net water use for oil extraction is calculated by subtracting the produced water injected from the produced water category in the total water injected.

Net water use = Total water injected by type – produced water injected (Equation A.3.1)

For each year and for each field, the *Total* and *Net* water use intensity (abbreviated below as **T**WUI and **N**WUI respectively) of secondary oil extraction is calculated as:

Method 1: TWUI or NWUI = Total or Net water injected / Total oil extracted (Equation A.3.2)

Method 2: Energy NWUI = Net water use / Total oil and natural gas extracted (Equation A.3.3)

where water is measured in liters, and oil is measured either in volumetric (liters) or energy (MJ) units. The energy density of crude oil is taken as 6.19 GJ per L (by LHV) based on a lower heating value of 5.8 MMBTU/BBL or 1055 MJ/MMBTU. The lower heating value for gas is taken to be 1.023 MMBtu/Mcf and 1055 MJ/MMBtu (all values are taken from the EIA AEO 2011, Appendix A: British Thermal Unit Conversion Factors).

Total and Net Water Use Intensity of Tertiary Oil Production

California oil wells use two types of tertiary technologies to produce oil: steam flooding and cyclic steam injection. In 2012, 70% of total tertiary water injected was used for steam flooding and 30% for cyclic steam injection. The water use intensity for these two technologies is calculated slightly differently. Both technologies use natural gas produced onsite or imported from elsewhere (if onsite production is insufficient) in cogeneration plants to generate steam that is then used to extract crude oil (Brandt 2011). Table 2 lists California's cogeneration facilities operating in 2012, together with the volumes of gas produced and required for oil production in 2012.

Table 2. Gas Required and Produced Onsite on Tertiary Fields in 2012.

Field Name	Steam Cogen capacity (gal/hr)	Gas Required (MCF)	Gas produced in 2012 (MCF)
Arroyo Grande	325,228	164,250	0
Belridge, North			
Belridge, South			
Cymric	6,697,952	4,234,000	3,644,772
Jasmin	7,255,093	NA	**Coal**
Kern Front	8,230,778	16,328,640	
Kern River	75076028	68,656,500	438
Lost Hills	1,651,159		
Lynch Canyon	0		
McKittrick	3,752,635	1,831,801	2,511,911
Midway-Sunset	55,900,398	42,340,000	4,874,346
Mount Poso	625,439	NA	**Coal**
Orcutt	0		
Coalinga	12,934,080		
Poso Creek	7,255,093	NA	**Coal**
Round Mountain	0		
Placerita	7,508,274	17,702,500	0
Oxnard	0		
San Ardo	13,414,782		

Source: (DOGGR 2013a, DOGGR 2013b)

To allocate water by the energy content of marketed energy products, the volume of natural gas required to generate steam given oil production is therefore first subtract the before allocating net water use to oil and gas produced. The amount of natural gas produced in each field in a given year is estimated as:

Net natural gas produced = natural gas produced by field - natural gas demand for steam production (Equation A.3.4)

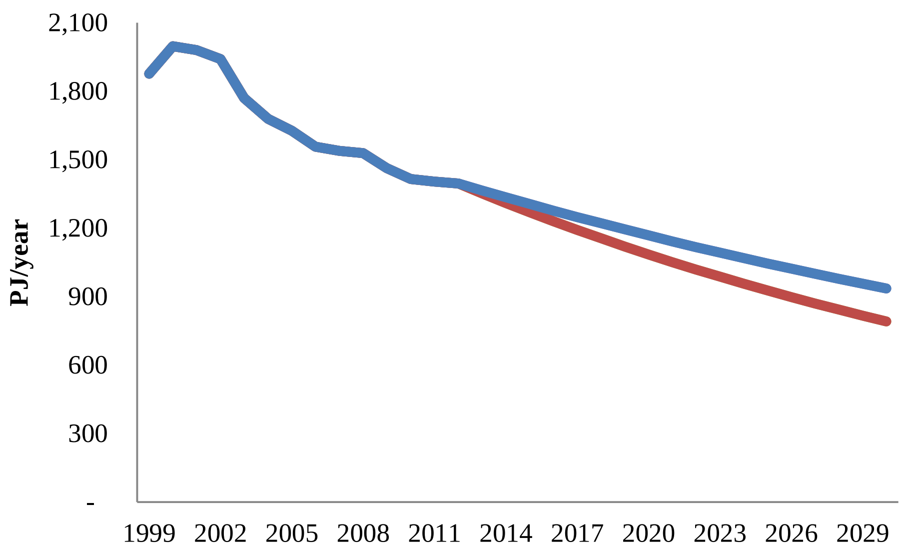
where all quantities are calculated in MJ. All but one of the seventeen tertiary fields did not produce enough natural gas onsite to generate the steam required for oil extraction in 2012. In this particular case, natural gas imported for energy extraction is neglected in calculating the WUI. The amount of gas produced is assumed to be zero in all the other cases. To calculate the WUI, the procedure outlined in *Method 2* (i.e. Equation A.4.3) is used to calculate the *Total* and *Net* WUI of tertiary oil extraction.

The net WUI range of California's tertiary production as calculated by *Method 1* and *Method 2* does not vary widely, as there are relatively few cogeneration facilities, and these use relatively small volumes of natural gas use on site. Relatively little net gas is produced at these fields, and as much is consumed in the steam making process either in cogeneration facilities as in once through steam generators (OTSGs). In previous years, one field did not have the cogeneration capabilities to use the natural gas produced on-site (McKitterick 2005-2009, 2011-2012), and three other fields used coal generators rather than steam (Jasmin, Mount Poso and Poso Creek).

C.2 Projecting Total Water Injection by Field in 2030

Regressions are fit on a field-by-field level using two functional forms: untransformed Ordinary Least Squares (OLS) and OLS based on a log-transformation of gross injected water volumes, against field age as the single independent variable. Taking the linear/log transformed *Total* WUI as the response variable, field age and type as fixed effects, and field name as a random intercept with varying slope. Using the projected *Total* WUI the type of water use was considered and used this to calculate the *Net* WUI based on the 2012 source breakdown. The gross and net water use in 2030 then calculated, using CEC projections of California oil production (2.1% and 3.2% annual decline, respectively) applied to each field, as shown in Figure 2.

Figure 2. California's Crude Oil Production in the Reference & Deep-GHG Scenarios.



Using a stepwise deletion approach from the maximal model, the minimal adequate model was selected which consisted of significant terms assessed by residual deviances to a chi-square distribution with residual degrees of freedom. Variables including field depth, which was previously found to be a significant variable in the literature for steam flooding and cyclic steam wells (Brandt 2011, El-Houjeiri, Brandt et al. 2013) and at Midway-Sunset first purchase price (from EIA data tables)[1] were found to be not significant and were removed from the minimal adequate model (Table 3).

Table 3. Mixed-Effects Model for Total Water Use Intensity.

$$\log(1 + TWUI) = FieldAge + FieldType + random(1 + FieldAge|FieldName)$$

Random effects:

Groups	**Name**	**Variance**	**Std. Dev.**
Field Name	(Intercept)	17.660	4.202
	Field age	0.003	0.059
Residual		0.064	0.253

Number of obs: 983, groups: 88

Fixed effects

Parameter	Estimate	Std. Error	t value
(Intercept)	-0.160	0.500	-0.320
Field Age	0.022**	0.007	3.151
Field Type Secondary	0.107	0.097	1.105
Field Type Tertiary	-0.170*	0.071	-2.395

Significance codes * p<0.05, ** p<0.01, *** p<0.001

In the random effects functional form there was a good deal of variability in both the intercept and the relation with <u>*Field Age*</u> variable. In the fixed effects form, <u>*Field Age*</u> was positively correlated with total WUI. As <u>*Field Age*</u> increased, total WUI logarithmically increased by a coefficient of 0.022. <u>*Field Type*</u> was also significant in the model, with secondary fields having a greater estimated total WUI, and tertiary and mixed fields having a lower derived total WUI. The AIC was the lowest of models calculated, indicating that the model had the best trade off between goodness of fit and model complexity. Table 5 shows the best-fit models (i.e. whether linear or log form was selected), by field, as well as the field-level projected total water use intensity (TWUI).

The model in Table 3 and the projected rate of oil production decline from CEC (2013) was used to estimate the total water use by field in 2030, absent other information, assuming that fields would continue to produce oil using the same extraction technology as in 2012. While freshwater use has declined in secondary production in favor of produced water re-use, freshwater use in tertiary fields has been increasing. In order to estimate the water type used in the *Baseline* water management scenario, it was assumed that fields would use the same proportion of water type that they did in 2012. However, similar gains in technology may occur in tertiary production, allowing for what may become essentially closed loop systems.

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[1] See EIA monthly time series data for the California Midway-Sunset first purchase price, available at: http://www.eia.gov/dnav/pet/hist/LeafHandler.ashx?n=PET&s=F005006143&f=M

Using the methodology and assumptions outlined above, the projected water injection in 2030 by type is shown in Table 4. The estimates suggest that oil production in 2030 would result in gross water use of 487 billion liters of water in the *Reference* GHG scenario (with a 2.1% annual decline in production) and 397 billion liters of water use in the *Deep GHG* production scenario.

Table 4. Water Use by Water Type and Production Technology in 2030.

Scenario	Field Type	Water Type				
		Other	**Produced**	**Fresh**	**Waste**	**Recycled**
Reference - *baseline* water						
	Secondary	10	176	7	-	-
	Tertiary	18	23	4	-	-
	Mixed	66	27	1	-	-
Reference - *water smart*						
	Secondary	10	176	3	-	3
	Tertiary	18	23	2	-	2
	Mixed	66	27	1	-	1
Deep GHG - *baseline* water						
	Secondary	8	148	5	0	-
	Tertiary	15	20	3	0	-
	Mixed	56	23	1	0	-
Deep GHG - *water smart*						
	Secondary	8	144	3	0	3
	Tertiary	15	19	1	0	1
	Mixed	54	22	1	0	1

Units are billion liters per year (projected for 2030).

Table 5. Best fit by Field, and Total WUI Prediction in 2030.

Field name	Best Fit	TWUI 2030
Aliso Canyon	log	0.6
Ant Hill	log	0.0
Antelope Hills, North	linear	10.2
Arroyo Grande	linear	11.1
Asphalto	linear	0.1
Bardsdale	linear	0.2
Belgian Anticline	linear	6.7
Belridge, North	linear	23.6
Belridge, South	linear	17.1
Beverly Hills	linear	23.1
Brea-Olinda	linear	7.6
Buena Vista	linear	71.2
Canfield Ranch	log	0.0
Cascade	linear	0.9
Castaic Hills	log	0.1
Cat Canyon	linear	0.9
Chico-Martinez	linear	9.5
Coalinga	log	6.9
Coles Levee, North	log	8.3
Coles Levee, South	log	0.2
Comanche Point	linear	10.0
Coyote, East	linear	51.8
Cuyama, South	log	80.8
Cymric	linear	6.8
Edison	linear	2.6
Edison, Northeast	linear	16.1
Elk Hills	linear	9.0
Fruitvale	log	0.0
Hasley Canyon	log	0.0
Huntington Beach	linear	50.5
Inglewood	linear	68.5
Jacalitos	log	0.1
Kern Front	linear	4.6
Kern River	linear	0.7
Kettleman North Dome	linear	2.2
Landslide	linear	68.4
Las Cienegas	linear	34.9
Long Beach	linear	30.6
Los Angeles Downtown	linear	34.9
Los Lobos	log	0.0
Lost Hills	linear	21.7
Lynch Canyon	log	0.1
Mahala	linear	0.2
McDonald Anticline	log	0.0
McKittrick	linear	4.4
Midway-Sunset	linear	8.3
Montalvo, West	linear	3.4
Montebello	linear	124.9
Mount Poso	linear	0.7
Mountain View	linear	3.8
Newhall	linear	3.1
Newport, West	log	0.5
Oak Canyon	linear	1.2
Oakridge	log	11.1
Orcutt	log	13.1
Oxnard	linear	3.2
Placerita	linear	13.8
Playa Del Rey	log	0.0
Poso Creek	linear	8.3
Pyramid Hills	linear	6.8
Richfield	log	22.0
Rincon	log	2.5
Rio Bravo	linear	29.3
Rosecrans	linear	35.9
Round Mountain	linear	5.2
Russell Ranch	linear	38.0
San Ardo	linear	10.5
San Miguelito	log	21.3
San Vicente	linear	6.2
Sansinena	linear	3.2
Santa Fe Springs	linear	66.7
Santa Maria Valley	log	46.6
Saticoy	log	0.3
Sawtelle	linear	12.2
Seal Beach	linear	7.8
South Mountain	linear	0.7
Tapia	linear	8.6
Tejon	log	0.0
Tejon Hills	linear	8.8
Tejon, North	linear	2.2
Torrance	log	12.8
Ventura	linear	12.5
Wheeler Ridge	log	0.4
Whittier	log	0.0
Wilmington	linear	69.2
Yorba Linda (ABD)	linear	76.9
Yowlumne	linear	38.4

Total WUI in liters water / liters oil produced.

C.3 Water Quality Impacts of Fracking in California

Chemicals used in fracking operations are reported voluntarily by some operators and documented in the *FracFocus* database. The majority of the chemicals reported in the database for fracking in California are of low toxicity or non-toxic. However, some chemicals of concern were identified, including biocides (e.g. tetrakis(hydroxymethyl)phosphonium sulfate; 2,2-dibromo-3-nitrilopropionamide; and glutaraldehyde), corrosion inhibitors (e.g. propargyl alcohol), and mineral acids (e.g. hydrofluoric acid and hydrochloric acid) (Jaffe 2014).

Groundwater contamination may occur through possible underground leakage from the wellbore to drinking water aquifers and improper disposal or accidental leakage of fracking fluids to surface water bodies. As the depth of most shale plays greatly exceeds that of groundwater aquifers, thousands of feet of rock separate fractures within the shale and drinking water aquifers, making it unlikely that fracking fluids leak underground, provided that there are no leaks in the wellhead or well casing (Clark, Burnham et al. 2012). However, shallower shale deposits may be vulnerable to this connection, such as Pavilion, Wyoming, where as little as 400 feet separates gas deposits from drinking water resources. The relatively high porosity of California's geological formations increases the likelihood of water migration (Pepino 2014).

As discussed in the main body of the report, one wastewater management strategy currently implemented in California and Wyoming is the reuse of diluted treated hydraulic fracturing wastewater for beneficial uses, including crop irrigation. However, the treated water has not yet been studied with respect to its uptake in agricultural crops. Arsenic was found in treated flowback water applied to agriculture, and has been shown to bioaccumulate in rice plants (Shariq 2013). Organic hydrocarbons have also been found in wheat plants grown in contaminated soil (Tao, Zhang et al. 2009).

Table 6 shows the number of unique wells using a common treatment as identified by *FracFocus*. Wells may use multiple chemicals that serve the functions listed below. Only seven wells (or one percent of the wells in California) used hydrochloric acid, or "acid treatment" to help dissolve minerals and initiate cracks in the rock, compared to 11,084 (66%) of fracking wells in the rest of the U.S. Thirteen percent of California wells used biocides and 99 percent of California fracking wells used gelling agents such as methanol, shown to cause neurological damage and dermatitis in the case of acute toxicity (Colborn, Kwiatkowski et al. 2011). In the rest of the U.S. 60 percent of fracking wells used biocides, and 96 percent of wells used chemicals identified as gelling agents by *FracFocus*.

Acid treatment typically uses hydrochloric acid to dissolve minerals and initiate cracks in the rock matrix containing the oil/gas deposit. Biocides (such as tetrakis hydroxymethyl-phosphonium sulfate and glutaraldehyde) are used to eliminate bacteria in the water that can produce corrosive by-products such as gases (particularly hydrogen sulfide), which could contaminate methane gas. Biocides also prevent the growth of bacteria, which can reduce the ability of the fluid to carry proppant (often sand) into the fractures. Breakers, such as sodium chloride and magnesium peroxide, delay the chemical break down of the gel. Clay stabilizers are used to prevent clays from swelling or shifting. Corrosion inhibitors act as product stabilizers or winterizing agents, or are used to prevent the corrosion of the pipe. Crosslinkers are used to maintain fluid viscosity as temperature increases. The chemicals under the product category of friction reducer "stick" to the water to minimize friction or act as winterizing agents.

Table 6. Percent of Fracking Wells Using Chemical Products in California & U.S.

Product Function	California	Rest of U.S.
Acid	2%	66%
Biocide	13%	60%
Breaker	86%	70%
Clay Stabilizer	6%	59%
Corrosion Inhibitor	28%	84%
Crosslinker	100%	95%
Friction Reducer	99%	93%
Gelling Agent	99%	96%
Iron Control	16%	55%
Non-Emulsifier	81%	67%
pH Adjusting Agent	99%	71%
Scale Inhibitor	100%	13%
Surfactant	30%	91%

All data are from *FracFocus*.

Gelling agents, used in over 95% of wells in California and the rest of the United States, include chemicals that thicken water to suspend sand (guar gum or polysaccharide blend) and product stabilizers. Citric acid, acetic acid, and thioglycolic acid are common iron control agents, used to prevent precipitation of metal oxides, carbonates, and sulfates (e.g. calcium carbonate, calcium sulfate, barium sulfate), which could plug off the formation. Non-emulsifiers, such as lauryl sulfate, are used to prevent the formation of emulsions in the fracture fluid. pH adjusting agents are used to control the pH of fluid to maintain the effectiveness of other components, such as crosslinkers. Scale inhibitors, such as sodium polycarboxylate, prevent scale deposits in the pipe. Surfactants are used to increase the viscosity of the fracture fluid. Some chemicals, such as methanol, are used for multiple purposes – methanol acts as a corrosion inhibitor, crosslinker, and a gelling agent.

Appendix D. Biofuels – Scenarios, Data Sources, and Assumptions

This appendix briefly describes the policy scenarios, economic model, and geographic (survey and satellite) data sources used to estimate and project water use for biofuel production at a national level and in California in 2030.

D.1 National Biofuels Policy Scenarios – the M-RFS22 and a National N-LCFS

The *Renewable Fuels Standard* (RFS1) was a key provision of the Energy Policy Act (2005). It envisioned volumetric mandates for domestic biofuels, with program design to be directed by the U.S. Environmental Protection Agency (EPA) in cooperation with the Departments of Energy and Agriculture (DOE, DOA). Under the Energy Independence and Security Act (2007), the EPA specified four renewable fuel categories, designated certain production pathways as belonging to one of the four categories, and allowed for adding more pathways in the future. In 2009, the EPA proposed increasing the original volumetric quotas. These and other proposed amendments were codified in the Independence and Security Act of 2007 were amended in the Renewable Fuel Standard Program Regulatory Impact Analysis (RFS2) (EPA 2010).

In February and again in June 2013 (the latter pending finalization), The EPA again substantively altered the RFS2 program, issuing a Notice of Proposed Rulemaking expanding the list of feedstocks that can be classified under the cellulosic/advanced pathways and added advanced butanol and electricity generated from landfill biogas. Renewable gasoline and diesel, renewable naphtha, butanol with a >50% reduction in GHG from the 2005 baseline, and electricity and renewable CNG/LNG from landfill biogas are now considered cellulosic/advanced biofuels. This legislation is interpreted as a way to address the rising costs of Renewable Identification Numbers (RINs) driven by rising quotas for second-generation and advanced biofuels.

For the analysis of the domestic and in-state water use implications of National and statewide biofuels policies, it should be emphasized California is not currently and will not likely in the timeframe of this analysis (to 2030) be a major site for production of biofuel feedstocks on agriculture lands. For this reason, a national scope is adopted, and the impacts of the modified RFS2 (M-RFS2) and a hypothetical National Low Carbon Fuel Standard (LCFS), which has gained intermittent attention from the Executive branch and Congress due to the relative success and more market-friendly design of the California N-LCFS vis-à-vis the RFS2. The RFS2 is contrasted with a scenario with both the RFS2 and National N-LCFS, and finally provide a counterfactual (BAU) scenario to explore what might be expected to occur in the absence of any national biofuel promotion policy.

1. BAU: The 'business-as-usual' scenario is best thought of as a counterfactual, without any government policies promoting biofuels. No policy is imposed; 2007 base year inputs match actual cropping patterns, but the RFS2 blender's tax credit is discontinued immediately.

2. M-RFS: The RFS2 (H.R. 3097, 2011) specifies volumetric mandate for each type of fuel year by year to 2022, as modified under technological advancement targets set out in the EIA's Annual Energy Outlook, 2010. It calls for a steady expansion in production of corn-based ethanol, but with a cap of 15 billion gallons per year starting from 2015. By 2022, the total volume of biofuels is to reach 36 billion gallons, of which no more than 15 billion gallons should be sourced from corn starch derived ethanol, 16 billion gallons from cellulosic ethanol, and 5 billion gallons from other advanced biofuels. Note that the M-RFS22 policy as modeled in the version of BEPAM used here includes the volumetric ethanol excise tax credit for blending ethanol with gasoline as extending through the final year of the analysis, 2030. In fact the tax credit expired at the end of 2011. Both RFS scenarios also include a cellulosic Biofuel Producers Tax Credit (CBPTC) to fuel blenders, a tax credit that is uniformly valid across all biofuels produced from cellulosic feedstocks. The M-RFS2 volumetric mandate requirement peaks in 2030 at 37.5 billion liters.

3. N-LCFS + RFS2: A National Low Carbon Fuel Standard (LCFS) is implemented at a stringency that leads to the same level of cumulative carbon emission reductions as under the RFS, through 2030. The N-LCFS incentivizes biofuel feedstocks and production pathways according to their relative carbon intensity. Effectively it translates to an implicit subsidy for biofuels in proportion to the inverse magnitude of their carbon intensity, and an implicit tax on gasoline and diesel fuels (also in proportion to their carbon intensity). Note that the estimated carbon intensity of biofuel production pathways in the N-LCFS scenario modeled here does include an 'ILUC factor,' meant to penalize feedstocks according to their estimated land use change impacts.

D.2 The BEPAM Model Structure, Inputs & Outputs, Assumptions, & Limitations

This study relies upon the *Biofuel and Environmental Policy Analysis Model* (BEPAM) to model the economic and land use implications of three policy cases from 2007-2035. BEPAM was developed by a team of researchers working under Professor Madhu Khanna, at the University of Illinois at Urbana-Champaign. Since the BEPAM assumptions and outputs form the basis of the water use analysis, a brief description of the most relevant assumptions, parameters, and results is provided here. For a complete list of peer-reviewed publications and policy reports using BEPAM, see: (Chen 2010, Huang, Khanna et al. 2011, Khanna, Chen et al. 2011, Chen, Huang et al. 2012, Chen, Khanna et al. 2012, Chen and Onal 2012, Khanna and Zilberman 2012).

BEPAM was designed to analyze quantitative environmental, economic, and energy impacts of U.S. domestic biofuels policies. It is a stylized dynamic multi-market, multi-period, price-endogenous, nonlinear mathematical programming model of the food/feed and fuel sectors. Geographic resolution and scope are highly detailed for the contiguous U.S., with base year calibrated yield resolution at the county level (and sensitive to crop management including irrigation and tillage practices) and variable acreage resolution at the Crop Reporting District (CRD: an aggregation of typically 6-10 counties and is the basic unit of agricultural census data aggregation). In addition to detailed modeling of the agricultural and transportation fuel sectors in the U.S., BEPAM contains basic linkages to global markets in both sectors, as estimated in aggregate for the rest of the world (ROW). Linear demand functions characterize homogenous consumer preferences for all commodities/services, including crop and livestock products and vehicle miles (VKT). Crop and livestock products can either be consumed domestically or else traded imported/exported with the ROW, as modeled by linear supply/demand functions, which are shifted upward over time at exogenously specified rates, reflecting increasing demands due to the population and income growth.

Policy impacts on prices and volumes result in annual endogenous determination of "(1) commodity prices; (2) production, consumption, export and import quantities of crop and livestock commodities and fuel and biofuels; (3) land allocations and choice of practices (namely, rotation, tillage and irrigation options) for producing row crops and perennial crops." (2010). Also the model endogenously determines annual greenhouse gas (GHG) emissions, in units of GWP_{100}, by aggregating across carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O).[2]

The BEPAM model makes certain stylized modeling assumptions. In particular, transportation demand takes place within an open economy (i.e. no taxes on petroleum or fuel imports), with homogeneous consumers who demand exogenously increasing levels of vehicle kilometers traveled (VKT) as projected by the Energy Information Administrations Annual Energy Outlook, 2010 (2010). Transportation fuels are produced by blending gasoline and biofuels as perfect substitute, that is, no restriction on aggregate volumes of ethanol or biodiesel result from current blendwall limitations. In other words ICE vehicles are assumed to either accommodate E15 blends or a transition to E85 blends is implicitly assumed. In addition to domestically produced biofuels, BEPAM considers biofuel imports from Brazil and Caribbean Basin Initiative (CBI) countries, albeit CBI biofuel imports are capped by an import quota. For biomass conversion, BEPAM assumes a conversion factor of 87.3 gallons of ethanol per metric tonne and 47.386 gallons of BTL per metric ton of cellulosic feedstock, independent on feedstock type.

For the purposes of land use and agricultural modeling, the CRD is taken as the decision-making unit. As outlined above, crop yields, the costs of crop and livestock production, land availability and GHG emissions differ across CRDs. As they are expected to constitute a large proportion of the total production costs of dedicated biofuel feedstocks on cropland, opportunity costs of land are explicitly tracked and endogenously modeled. These costs are measured as the difference between the per land area revenues and production costs from the most profitable crop production practice (such as crop rotation and tillage).

Certain land types, including arable land set aside for the Conservation Reserve Program (CRP), forest land, and pasture land, are not available for either row crop or dedicated biofuel feedstock (i.e. switchgrass and miscanthus) production, as CRP land has been set aside to improve soils and improve ground and surface water quality. Other marginal lands are available for dedicated feedstock production; idle land and cropland pasture are taken as *marginal* lands capable of supporting these perennial grasses, though they are modeled as incurred a yield penalty of 66% of prime (cropped) land yields. It is assumed that dedicated feedstocks are not irrigated, as in most regions of the country it would be economically infeasible to do so.

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[2] In BEPAM, GREET lifecycle emission factors were used (Wang, M. (2009). "GREET 1.8 c." Argonne National Laboratory.). GREET 1.8c accounts for both direct and indirect emissions including those of fertilizers, pesticides, and machinery at the production/cultivation stage, as well as emissions incurred by production and transport of those inputs to the farm, of processing/transportation of the feedstock to a biorefinery; conversion to biofuel and transport of the biofuel for final consumption.

Crop and livestock production supply and land use competition are linked, and spatial heterogeneity in both activities is modeled using Leontief production functions and historical data for crop production costs, yields, and resource endowments. The model includes both primary livestock products (eggs and milk) and secondary commodities (oils from, corn, soybeans, and peanuts; other secondary crop products: soybean meal, refined sugar, high-fructose corn syrup (HFCS); wool; and meat from livestock: beef, pork, turkey, chicken and lamb).

'Historical' and 'synthetic' crop mixes are used to constrain future cropping patterns while simultaneously allowing switchgrass and miscanthus to be cultivated according to their suitability. As new price regimes are projected in the coming 25 years, BEPAM allows crop acreage to change by up to 10% each year from observed cropping patterns within each CRD. A crop productivity model, *MISCANMOD*, is used to simulate the yields of miscanthus and switchgrass. The *MISCANMOD* model estimates yields of miscanthus and two varieties of Cave-in-Rock switchgrass (upland and lowland varieties) using GIS data on climate, soil moisture, solar radiation and growing degree days as model inputs, as described in Jain, Khanna et al. (2010). Other BEPAM model runs have tested the sensitivity of results to other specifications of miscanthus and switchgrass productivity according to climatic and soil inputs.

Although BEPAM does not explicitly incorporate water use limitations into its scenarios for dedicated perennial grass land use expansion, it does impose a restriction of 25% of total available land on miscanthus production, citing the potential detrimental impacts of monocropping of a non-native grass over large land areas on biodiversity and water quality.

D.3 BEPAM Data Description

The BEPAM model considered 22 agricultural commodities, which are traded with the ROW. These are produced across 295 CRDs. Agricultural data relevant to estimating water impacts consist of *five land types*: cropland, idle cropland, cropland pasture, pasture land, and forestland pasture. Area protected by the Conservation Reserve Program (CRP) are off-limits for cropping by row crops and biofuels feedstocks.

Cropland is synonymous with currently cropped land, and is taken as prime land. As explained above, idle cropland and cropland pasture are available for switchgrass and miscanthus cultivation, albeit with an associated yield penalty (0.66) as these are considered marginal lands. Note that there is no matching classification between these land types and the satellite-derived land use classifications mapped in the *Cropland Data Layer* (described in Section A.5.3) (Han, Yang et al. 2012). Crops modeled in BEPAM include *two bioenergy crops:* switchgrass and miscanthus; and *twelve row crops:* corn, soybeans, wheat (***durum, winter***, spring), rice, sorghum, oats (spring, ***fall***), barley (spring, ***fall***), cotton, ***peanuts***, sugarbeets, sugarcane, corn silage.

The following *feedstocks* are available for conversion into biofuel products (indicated in parentheses): corn and sugarcane (conventional ethanol); soybeans (biodiesel); corn stover, wheat straw, forest residues, miscanthus, switchgrass, and imported sugarcane (cellulosic ethanol).

Note that the water use impacts of forest residues and sugarcane are outside the scope of this analysis, as their production in BEPAM is not assumed to have any domestic land use or water use impacts (i.e. the removal of forest residues has no associated LUC effects and the water use impacts are considered negligible, and sugarcane production only occurs abroad). Table 7 shows the row crops and energy crops considered in the analysis, as well as the national aggregate yields, under irrigated and rainfed production, as modeled by BEPAM, and the national low/high mean national yield ranges for these same crops. An exogenous multiplier averaging 1.173% is applied in BEPAM to model the annual increase in crop yields, across all crops, from 2007-2035.

Land use data across non-cropped land types (idle cropland, cropland pasture, pasture and forestland pasture) at CRD level resolution come from the most recent detailed national agricultural census (NASS 2007). Aside from land protected by the CRP, both idle cropland and cropland pasture are available for biofuel feedstock cultivation; the total availability of idle cropland is 15 M ha while cropland pasture is 13 M ha. Marginal lands of these two classes may be converted to dedicated biofuel feedstock cultivation in response to the changing crop prices.

The model was validated for the base year by comparing results for land allocation at CRD level resolution with 2007 NASS census data. Crop and commodity prices were also compared and were in most instances within 10% of census reported values, while fuel prices and consumption differed by less than 5% from their actual values (Chen, Khanna et al. 2012). Cost data (e.g. for fertilizer, pre-harvesting, and harvesting) come from the 2009 NASS surveys. Costs of miscanthus and switchgrass production, and import/export supply and demand elasticities are taken from relevant literature sources. The reader is referred to relevant reports on GCAM (Chen 2010, Khanna, Chen et al. 2011, Chen, Huang et al. 2012, Chen, Huang et al. 2012, Chen and Khanna 2012, Chen, Khanna et al. 2012, Chen and Onal 2012, Chen, Huang et al. 2014).

Table 7. National Average Yields for Crops Modeled in BEPAM.

Crops	*NASS 2008 dates*	*NASS 2008 (FRIS)*	*BEPAM range - dryland*			*BEPAM range - irrigated*			*National average NASS yields*		*UNITS (DM yield)*
	planting/ harvesting	irrigation data	*min*	*mean*	*max*	*min*	*mean*	*max*	lower	upper	
Oats, fall	X	X - area	8.9	59.6	120.0	53.0	83.6	114.4	57.1	67.5	metric tons/acre
Oats, spring	X	X - area	8.9	59.6	120.0	53.0	83.6	114.4	57.1	67.5	metric tons/acre
Corn	X	X	26.7	122.0	220.2	31.6	164.9	205.0	147	164	metric tons/acre
Soy	X	X	6.2	36.2	62.8	23.0	49.0	64.4	39.7	44	metric tons/acre
Wheat, spring	X	X	5.0	47.7	115.1	16.0	60.9	117.9	38.6	46.3	metric tons/acre
Wheat, winter	X	X	5.0	47.7	115.1	16.0	60.9	117.9	38.6	46.3	metric tons/acre
Wheat, durum	X	X	5.0	47.7	115.1	16.0	60.9	117.9	38.6	46.3	metric tons/acre
Sorghum	X	X	11.0	63.5	127.0	18.2	87.0	135.0	54.6	73.2	metric tons/acre
Barley, fall	X	X	10.0	61.1	146.2	15.0	84.2	150.0	60	73.1	metric tons/acre
Barley, spring	X	X	10.0	61.1	146.2	15.0	84.2	150.0	60	73.1	metric tons/acre
Corn stover	X		0.1	1.8	2.5	0.8	1.6	2.3	*2.0*	*2.3*	metric tons/acre
Wheat straw	X		0.1	1.4	1.6	0.8	1.2	1.7	*1.1*	*1.3*	metric tons/acre
Switchgrass			0.6	12.1	20.3	*NA*	*NA*	*NA*	*6.0*	*20.0*	metric tons/acre
Miscanthus			4.5	34.4	71.5	*NA*	*NA*	*NA*	*10.0*	*30.0*	metric tons/acre
									** literature cited ranges for cellulosic feedstocks*		
Cotton	X	X	570.0	982.0	1371.0	109.0	738.5	1594.0	772	879	metric tons/acre
Peanut	X	X - area	630.0	2905.0	4263.0	2375.0	3266.0	4588.0	2863	3426	metric tons/acre
Rice	X	X	4870.0	6606.0	8828.0	*NA*	*NA*	*NA*	6725	7219	metric tons/acre
Silage	X - harvesting only	X	3.9	15.9	33.0	5.0	20.2	33.0	16.2	19.3	metric tons/acre
Sugarbeet	X	X	15.0	24.2	43.4	20.7	28.2	41.2	23.7	27.7	metric tons/acre
Alfalfa/hay	X - harvesting only	X	0.6	2.5	8.4	1.1	2.7	4.8	2.1	2.3	metric tons/acre

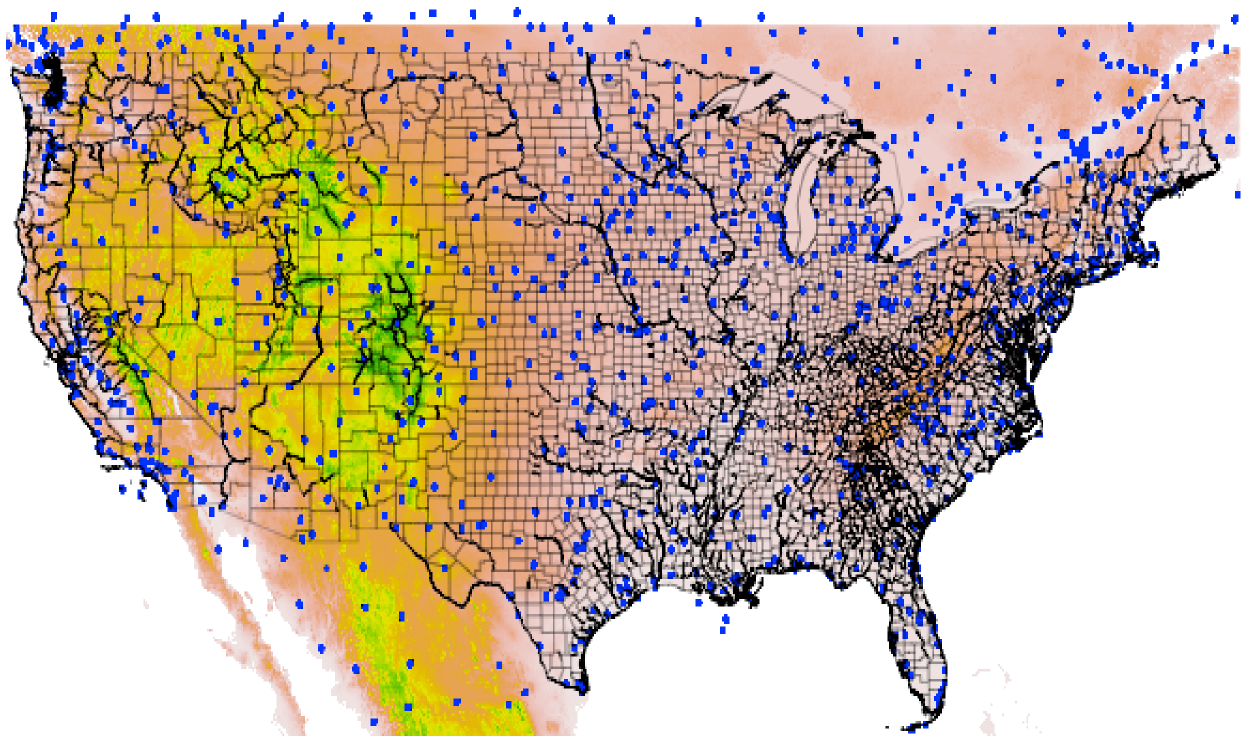
Sources: BEPAM model (2011 version), NASS (2010), Schaible (2013)

D.4 Spatial Data

Daily weather and solar radiation, altitude and soil data

Daily weather data are from NCAR (accessed 9.14.2013). Daily data from 1728 meteorological stations from 1994 to present were cleaned and interpolated by thin plate splines interpolation to the ~10 km² resolution in the same projection as the master level cleaned *Cropscape* (Han, Yang et al. 2012) cropping pattern data. Stations contained locational data (latitude, longitude, elevation), as well as daily maximum/minimum temperature, precipitation, vapor pressure, vapor pressure deficit, incident solar radiation, and eight 3-hourly wind speed readings were available for more than 87% of the daily readings. In cases where missing data series were no more than two days long, missing data were interpolated temporally. Station locations were confirmed and corrected in 11 instances with 30 m resolution altitude data from USGS (SRTM30 Version 2 accessed 9.10.2013). Figure 3 shows the meteorological station locations and altitude.

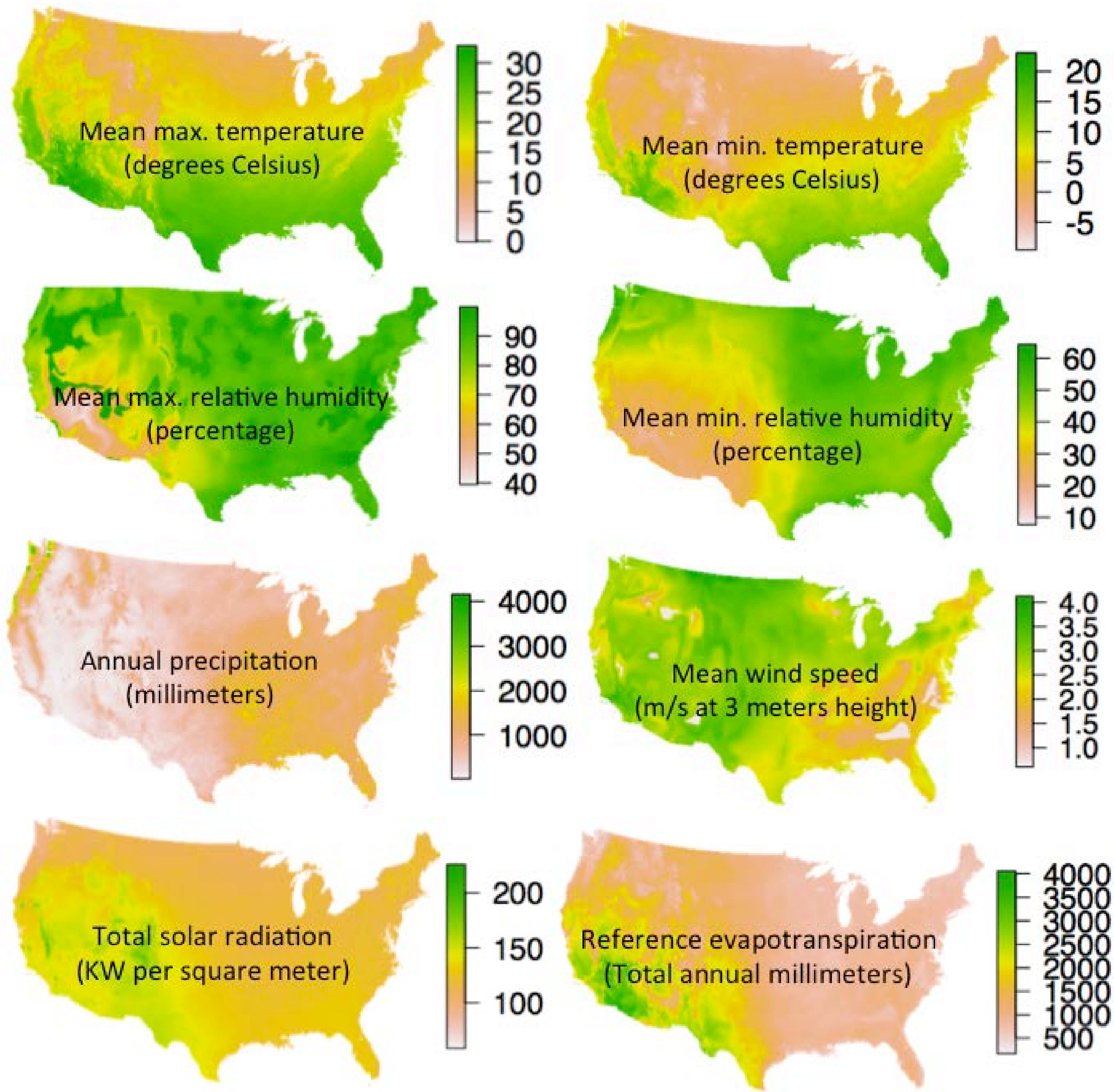
Figure 3. Meteorological Stations & Altitude for Derivation of Daily Weather Parameters.



Daily station weather parameters, together with station elevations, were used to derive daily reference evapotranspiration (ETo). Elevation mismatches were also used to correct incorrectly coded station latitude and longitude data.

Higher resolution and quality daily shortwave incident solar radiation data (1994-2012) was taken from Thornton, MM et al. (2012). Figure 4 summarizes the interpolated weather parameters at the 10 km spatial resolution. It shows total annual incident precipitation in 2007, as well as annual average minimum/maximum relative humidity and temperature. Finally it shows annual average ETo values, which were derived as detailed in the following section.

Figure 4. Annual Total / Average Values of Key Daily Weather Input Parameters.



Annual average (temperature, relative humidity, wind speed) / total (precipitation, solar radiation, ETo) values of the primary weather inputs used to derive reference evapotranspiration (ETo) in the base year. Units are annual means/totals of daily observations.

Calculation of Daily Reference Evapotranspiration

The reference evapotranspiration (ETo) is calculated at a daily timestep across all grid cells in the contiguous U.S. These calculations followed the methods outlined in Section 4 of FAO 56 (Allen, Pereira et al. 1998), by the Penman-Monteith equation:

$$\lambda ET = \frac{\Delta(R_n - G) + \rho_a c_p \frac{(e_s - e_a)}{r_a}}{\Delta + \gamma\left(1 + \frac{r_s}{r_a}\right)} \qquad \text{(Equation A.4.1)}$$

where R_n is the net radiation (and is a function of the three constants, and sky cover observations weighted by incoming solar radiation), G is the soil heat flux (negligible at daily timesteps), (e_s - e_a) represents the vapor pressure deficit of the air, ρ_a is the mean air density at constant pressure, c_p is the specific heat of the air, Δ represents the slope of the saturation vapour pressure temperature relation (and is a function of min/max temperature and min/max relative humidity), γ is the psychrometric constant, and r_s and r_a are the (bulk) surface and aerodynamic resistances (which are functions of wind speed). The calculation procedures for ETo , written in R, were verified with online open-source software using the same derivation algorithm. For further details on the derivation of daily ETo at ~10 km^2 across the entire U.S., see FAO 56, Chapter 4 (Han, Yang et al. 2012), and the source code.

Annual cropping patterns

A time series of satellite-derived classifications of historical cropping data from USDA's *CropScape* (Han, Yang et al. 2012) were extracted for each year from 1997-2012. From the original 255 classifications, 32 cropping patterns were selected that correspond to those crops and land types explicitly modeled in BEPAM. Table 8 shows the reclassification from original crop codes to modified cropping classifications given in BEPAM.

Data from 1997-2007 are restricted to a subset of states in the contiguous U.S. (1997 data are available for a single state, and by 2007, 21 states, including the main crop producing states, are covered). From 2008-2010 data are available across the lower 48 states. Original data, including double-cropped areas and crop rotations, were converted from 30/56 square meter resolution to approximately 25 square kilometer resolution. Aggregation techniques allow us to use estimate with a high degree of precision the number of hectares per 10-kilometer pixel cropped by any of the above listed cropping patterns.

Original data, including double-cropped areas and crop rotations, were converted from 30 or 56 square meter resolution to approximately 10 square kilometer resolution. Aggregation techniques allow us to use estimate with a high degree of precision the number of hectares per 10-km2 pixel cropped by any of the above listed cropping patterns. Cropscape began imaging some western states in 2008, for these regions, the 2007 data were merged with 2008, and were used to identify cropping practice locations for validation. Figure 5 shows aggregated land use as a percentage of total land, in the base year. All analyses below were performed in *R* unless otherwise noted. Source code is available upon request.

Table 8. NASS Classification and this Study's Reclassification of Satellite Land Use Data.

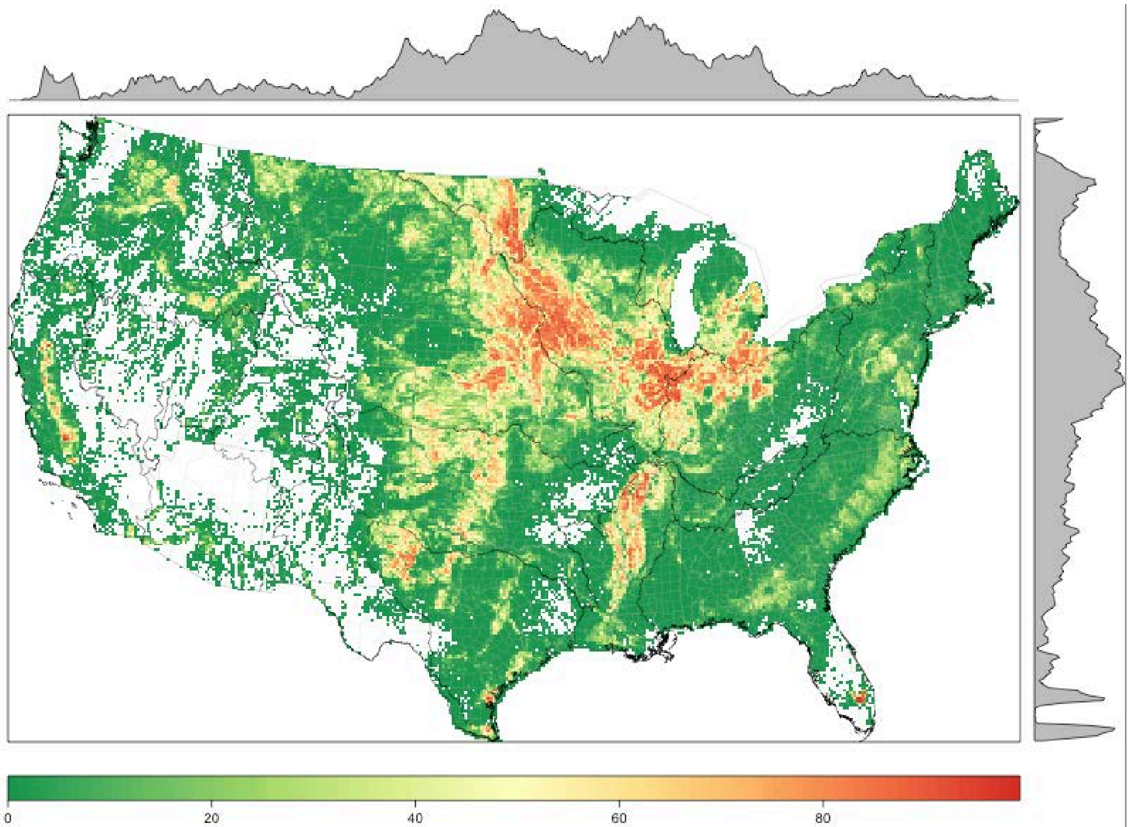
Code	*NASS classification*	*Reclassification*
1	Corn	Corn
2	Cotton	Cotton
3	Rice	Rice
4	Sorghum	Sorghum
5	Soybeans	Soybeans
10	Peanuts	Peanuts
12	Sweet Corn	Sweet Corn
13	Pop. or Orn. Corn	Pop. or Orn. Corn
21	Barley	Barley
22	Durum Wheat	Durum Wheat
23	Spring Wheat	Spring Wheat
24	Winter Wheat	Winter Wheat
26	Dbl. Crop WinWht/Soy	Dbl. Crop WinWht/Soy
28	Oats	Oats
36	Alfalfa/hay	Alfalfa/hay
37	Other Hay	Alfalfa/hay
41	Sugarbeets	Sugarbeets
45	Sugarcane	Sugarcane
60	Switchgrass	Switchgrass
61	Fallow/Idle Cropland	Fallow/Idle Cropland

Code	*NASS classification*	*Reclassification*
225	Dbl. Crop WinWht/Corn	Dbl. Crop WinWht/Corn
226	Dbl. Crop Oats/Corn	Dbl. Crop Oats/Corn
230	Dbl. Crop Lettuce/Durum Wht	Dbl. Crop Lettuce/Durum Wht
231	Dbl. Crop Lettuce/Cantaloupe	Dbl. Crop Lettuce/Cantaloupe
232	Dbl. Crop Lettuce/Upland Cotton	Dbl. Crop Lettuce/Upland Cotton
233	Dbl. Crop Lettuce/Barley	Dbl. Crop Lettuce/Barley
234	Dbl. Crop Durum Wht/Sorghum	Dbl. Crop Durum Wht/Sorghum
235	Dbl. Crop Barley/Sorghum	Dbl. Crop Barley/Sorghum
236	Dbl. Crop WinWht/Sorghum	Dbl. Crop WinWht/Sorghum
237	Dbl. Crop Barley/Corn	Dbl. Crop Barley/Corn
238	Dbl. Crop WinWht/Cotton	Dbl. Crop WinWht/Cotton
239	Dbl. Crop Soybeans/Cotton	Dbl. Crop Soybeans/Cotton
240	Dbl. Crop Soybeans/Oats	Dbl. Crop Soybeans/Oats
241	Dbl. Crop Corn/Soybeans	Dbl. Crop Corn/Soybeans
254	Dbl. Crop Barley/Soybeans	Dbl. Crop Barley/Soybeans
152	Shrubland	Crop_pasture
171	Grassland Herbaceous	Crop_pasture
181	Pasture/Hay	Crop_pasture

Code	*NASS classification*	*Reclassification*
63	Woodland	NA
64	Shrubland	NA
65	Barren	NA
82	Urban/Developed	NA
83	Water	NA
87	Wetlands	NA
88	Nonag/Undefined	NA
92	Aquaculture	NA
111	Open Water	NA
112	Perennial Ice/Snow	NA
121	Developed/Open Space	NA
122	Developed/Low Intensity	NA
123	Developed/Medium Intensity	NA
124	Developed/High Intensity	NA
131	Barren	NA
141	Deciduous Forest	NA
142	Evergreen Forest	NA
143	Mixed Forest	NA
190	Woody Wetlands	NA
195	Herbaceous Wetlands	NA

Original landcover and cropping patterns classifications from the National Agricultural Statistics Service (Han, Yang et al. 2012). Reclassification to match BEPAM crops, and marginal land types (in red) suitable for displacement by dedicated biofuel feedstocks (switchgrass & miscanthus).

Figure 5. Percentage of Land Cover Cropped in the Base Year.



Percentage of land cropped in modeled in the base year (2007). Cropping patterns are derived from the CropScape satellite for the subset of crops modeled in BEPAM. Patterns were taken from 2008 for states not covered in CropScape in 2007.

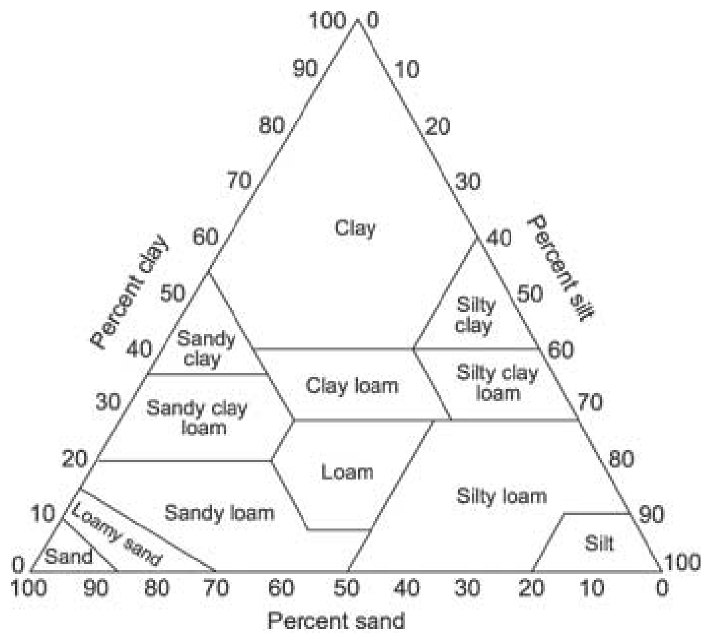
Although there is good agreement at the national resolution between the cropped areas reported in the NASS 2007 census data (and used in BEPAM) and the *CropScape* data, at the CRD resolution there are varying though minor degrees of mismatch between the total cropped areas, according to region and crop. CRD level areas from BEPAM were reweighted to match *CropScape* areas in the base year. In subsequent years, BEPAM derived areas for each cropping practice (crop sown, irrigation vs. rainfed, and no-till vs. conventional- vs. no-till--conventional rotations) were reweighted by the average hectares grown per ~10 km grid cell, and allocated proportionally to the average areas planted in *CropScape* over the 16 years of *CropScape* satellite monitoring.[3]

[3] One particular issue in allocating land arose from the fact that the acreage in satellite land patterns were categorized to correspond to BEPAM categories of 'idle cropland', 'cropland pasture', and 'pasture grass' were in some cases insufficient for allocation to switchgrass and miscanthus. In such cases, it was assumed that other lands were in deed available for expansion of dedicated feedstock cultivation onto marginal land.

Soil Data

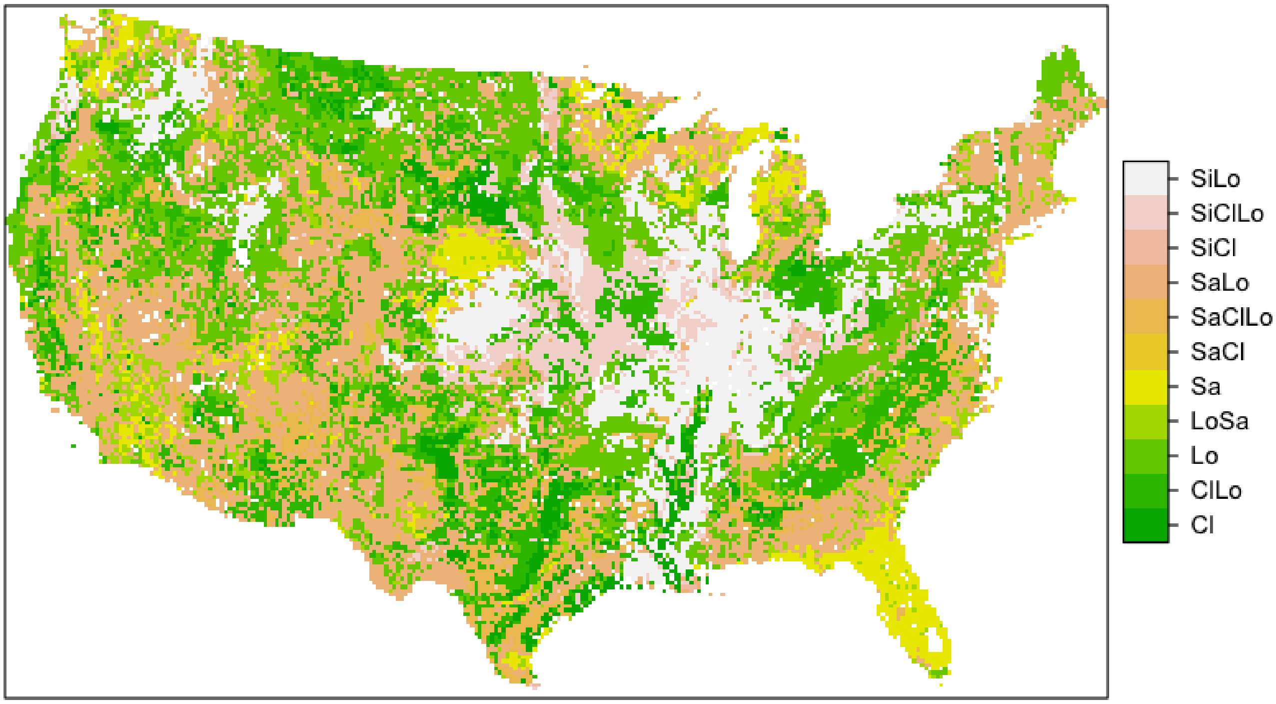
Soil data come from STATSGO (USDA accessed 6/9/2012). Texture data for the top 10, 30, 50, and 100 cm layer of soil were extracted, and the weighted-average of soil texture based on percentage sand, silt, and clay in the top four meters (or to the impermeable layer) of soil was used to derive soil texture class inputs for the daily calculation of water balances. Data was first cleaned to only include those regions with higher than 75% soil coverage (as opposed to rock, etc.). The soil profiles were then classified according to the USDA's soil texture classification triangle, shown in Figure 6.

Figure 6. The USDA Soil Texture Pyramid.



The resulting soil maps are shown in Figure 7. These classifications were used to estimate total evaporable water (TEW), and readily evaporable water (REW) for the soil-crop water balance calculations outlined in section A.5.5, below. Missing TEW and REW values for the less than 5% of ~10 km grid cells were interpolated using inverse distance weighting.

Figure 7. Soil Textures at 10 Kilometer Resolution across the Contiguous U.S.



Survey Data

Data on irrigation water volumes come from the NASS 2008 *Farm and Ranch Irrigation Survey* (FRIS) (Schaible 2013)[4], and from the USGS 2005 *National Water Use Database* (USGS 2005). The FRIS dataset includes disaggregated volume of water withdrawn by irrigation technology and mode of conveyance, as well as areal estimates of total irrigation cropland. These survey statistics are available at state-level resolution, by crop, for the majority of the crops modeled in BEPAM. The USGS provides total surface- and groundwater withdrawals for crop irrigation at a county level, albeit without disaggregating by conveyance/application technology, nor by crop.

For most of the crops modeled in BEPAM, planting and harvesting dates are given for 2009 at state-level resolution by NASS (2010). Daily water balance modeling used the median planting date and harvesting date for each state to define the growth see for each crop. State level planting and harvesting dates for switchgrass and miscanthus, and planting dates for silage and alfalfa/hay, were taken from an online survey of state agricultural service documents, and interpolated for states where no such resources were available.

[4] Available online at: http://www.agcensus.usda.gov/Publications/2007/Online_Highlights/Farm_and_Ranch_Irrigation_Survey/index.php

D.5 Parameterization According to FAO 56

Chapters 4 and 7-11 of FAO 56 (Allen, Pereira et al. 1998) detail the algorithms needed to calculate reference evapotranspiration, and daily crop water balances for all the cropping patterns modeled here, respectively. The calculations of water flows use the dual crop coefficient approach ($K_{cb} + K_e$) (Chapter 7), together with adjustments for conditions of water stress (Chapter 8). For irrigated cropping patterns as designated in BEPAM, irrigation scheduling under moderate water stress were implemented. For cropland pasture, idle cropland, and pasture grass, estimated parameter values for perennial, natural crops (e.g. growing seasons and stage lengths, rooting depths, K_{cb} values, etc.) are informed by Chapter 9. For fallow season estimation of weed transpiration and soil evaporation (as well as groundwater infiltration and runoff), the methods described in Chapter 11 of FAO 56 were adopted.

Use of the dual crop coefficient allows for separate estimation of plant transpiration and total evaporation, as well as daily runoff, groundwater irrigation, and irrigation. Water balances were aggregated for the growing season and fallow season. A separate document (forthcoming) details the methods and assumptions used to integrate those calculations with the geographic data, and execute them in *R*. The code used to execute this analysis is available on request.

Table 9 lists the key parameter values adopted to generate estimates of water balances in the base year and in 2030. For more information, see Appendix 5 and FAO 56 (Allen, Pereira et al. 1998).

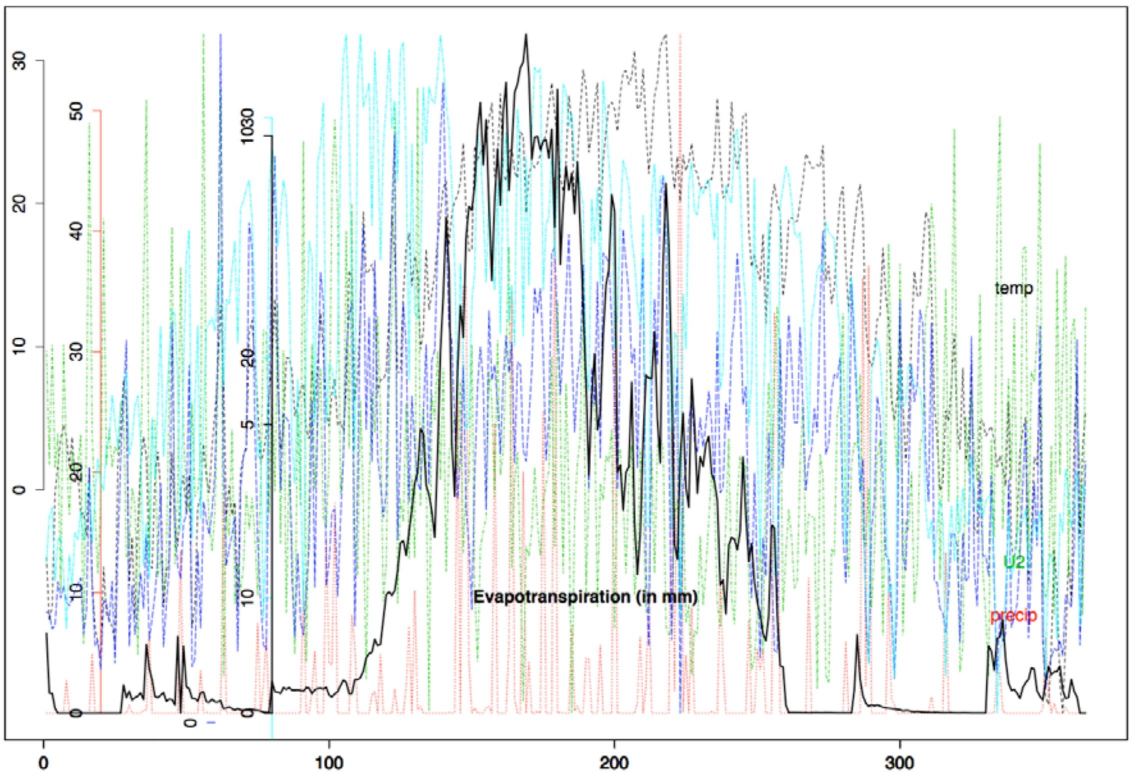
Table 9. Parameter Values Adopted for Crop-Water Modeling.

Parameter	File name	Description
K_{cb} values	*Kcb_values.csv*	The dual crop coefficient splits evapotranspiration into two terms: the basal crop coefficient (K_{cb}), and soil evaporation (K_e).
Plant rooting depths and p-values	*crop.roots.csv*	Crop soil water update is a function of effective rooting depth and p-value (soil water depletion fraction), a parameter that accounts for the efficiency of various crops' roots at absorbing water from dry soil.
Values adopted for soil water characteristics (TEW, REW)	*texture.TEW.REW.csv*	Depending on soil texture, soils have variable capabilities to hold, retain, and drain water. Total and readily evaporable and available water (TEW, REW, TAW, and RAW), are derived on the basis of soil texture (percent sand, silt, and clay).
Crop growth stage proportions (before state-level rescaling)	*Growth_stages.csv*	The relative length of the initial, mid, and late stages of crop growth is rescaled by the total length of the growing season (from planting through harvesting) at a state level.
State-by-state planting/harvesting dates (day of year)	*Planting_harvesting_dates_final.csv*	These data come from the NASS 2008 survey, or from state level agricultural extension services, or (for missing states) from interpolation based on the above two data sources.

The Kcb, rooting depth, and crop growth stage proportion values were calibrated by comparing the results with FAO 56 examples and with textbook values for various crop-water parameters.

Figure 8 shows a sample of certain key input parameters for sorghum grown in California in the base year. On the basis of detailed examination of input parameters like the sample shown here, it was determined that the model inputs matched the algorithms and made sense for the locations modeled.

Figure 8. Input Weather Parameters for Sorghum Grown in California.



Temperature (black) in degrees Celsius, wind speed (green) in meters per second at three meters height, precipitation (red) in millimeters, and derived modelled reference evapotransipiration (black) in mm.

Appendix E. Estimating the Water Use of Electricity Generation

E.1 Natural Gas Combined Cycle Power Plants (NGCC)

NGCC power plants are disaggregated into the following four categories, which differ in terms of water requirements:

(i) Plants with wet recirculating wet cooling (NG-CC-RC);
(ii) Plants with once-through cooling system (NG-CC-OT);
(iii) Co-generation plants (NG-CC-CG); and
(iv) Dry-cooled plants (NG-CC-DR).

NG-CC-RC

These 23 base-load plants contributed nearly a quarter of in-state electricity generation in 2005 (across all feedstocks and technologies). Most of these plants have wet recirculating cooling and use recycled water for cooling purposes. Further, most of these plants have Zero Liquid Discharge (ZLD) systems.

Table 10. Water Use Intensity and Sources for NGCC Plants.

	Macknick, Newmark et al. (2012)	**This Study**
Median WUI (liters/MWh)	776 (Tower); 908 (Cooling Pond)	240
Water Source		Recycled: 68%; Freshwater: 32%

All future NG-CC-RC power plants are assumed to use recycled water and be equipped with ZLD systems. Continued water scarcity and regulations will incentivize the use of recycled water. Similarly, environmental concerns will incentivize the use of ZLD facilities and prohibit discharge of blowdown to surface water bodies.

NG-CC-OT

According to the USC database, there are six NGCC plants with once-through cooling, two of which are sized at less than 50 MW capacity. These plants use ocean / estuary water for cooling. Except for two units of the Moss Landing Power Plant, all other plants are considered 'aging' by CEC (Vidaver, Ringer et al. 2009) and are currently operating at low capacity factors. As noted in the CEC report, "…the primary value of California's aging, gas-fired once-through-cooled units is capacity rather than generation, and they also make a significant contribution to local reliability…". Based on existing and evolving policies (Gregorio, Saiz et al. 2008), it is assumed that all OTC plants will be retired before 2030.

Co-generation Plants (NG-CC-CG)

The UCS report (Averyt, Fisher et al. 2011) reports around fifteen NGCC power plants which are used as cogeneration power plants and equipped with wet recirculating cooling towers.

The siting case for the BP Watson Cogeneration plant (a project for capacity enhancement from 385 MW to 470 MW) indicates that around three-quarters of the water requirements of such plants are for process steam for use by downstream industrial or mining activity. Given that heat remaining after electricity generation is transferred to produce process steam, the cooling water requirements are substantially lower than NG-CC-RC; the WUI of 151 liters/MWh is taken, based on BP Watson siting case.

No information is available about the other power plants. Water use by these plants is not reported in the EIA Forms 860/923. Similarly, Macknick, Newmark et al. (2012) and Averyt, Fisher et al. (2011) do not analyze cogen (CHP) plants separately and for such plants simply assume water use intensity of a typical NGCC plants with wet recirculating cooling.

Table 11. Water Use Intensity & Source Assumptions for NGCC Co-generation Plants.

	Macknick, Newmark et al. (2012)	This Study
Median WUI (liters/MWh)	776	151
Water Source		Not available - 100% freshwater assumed.

Most of the cogeneration plants in California are not grid-connected – while it was not possible to ascertain the exact breakup of CHP generated electricity that was locally consumed versus grid exported, analysis by ICF International for CEC (Hedman, Ken et al. 2012) indicates that most of the CHP capacity is dedicated towards local consumption and not exported to the grid. ICF analysis indicates that only a third of the total CHP capacity in 2011 has export potential – the actual capacity used for grid exports is likely to be lower than that. The CA-TIMES electricity projections exclude off-grid electricity. For simplicity, the same water use intensity was adopted for both NG-CC-RC and NG-CC-CG.

Dry Cooled Plants (NG-CC-DR)

There are only two dry-cooled NGCC power plants (not considering some of the smaller co-generation plants). While no water is required in the cooling towers, demineralized water is required for inlet fogging especially during peak summers and for boiler process water. Maulbetsch and DiFilippo (2006) estimated a water consumption intensity of 26-38 liters/MWh for inlet fogging, depending upon weather conditions (e.g. hot dry desert conditions, humid coastal conditions, etc.). The siting case for the Sutter Energy facility by Calpine indicates an annual average water consumption intensity of around 57 liters/MWh. Macknick, Newmark et al. (2012) estimate a much lower median water-use intensity of 8 liters/MWh based on a small sample of plants in the EIA database.

Assuming Zero Liquid Discharge systems (as is the case with Sutter Energy), water withdrawn is equal to water consumed.

Table 12. Assumptions for NGCC Dry-Cooled Plants.

	Macknick, Newmark et al. (2012)	Maulbetsch and DiFilippo (2006)	This Study
Median WUI (liters/MWh)	8 (Range: 0-15)	26-38 liters/MWh	38
Water Source			Freshwater: 100%

E.2 Concentrated Solar Power (CSP)

Water Use Intensity

Table 13 summarizes the median consumptive WUI of various combinations of solar thermal and cooling system technologies reported by Macknick, Newmark et al. (2012) based on an extensive literature review. The average water use intensities based on the seven approved solar thermal projects in California are included as well. Together, these projects have a combined capacity of 2,800 MW and 6.5 TWh electricity generation potential (assuming a 28% capacity factor). These projects are in various stages of construction and completion.

Table 13. Water Use Intensity of Concentrated Solar Power (CSP) Technologies.

	Wet Recirculating (RC)	Dry (DR)	Hybrid (HB)	Notes
Parabolic Trough (PT)	3,430	295	1,279	Analysis of CA approved projects indicates 2,725-2,775 liters/MWh for wet cooling; and 356-409 liters/MWh for dry cooling.
Linear Fresnel (LF)	3,785			
Power Tower (TP)	2,975	98	664	Both projects approved in CA have adopted dry cooling with a wide range of 106-322 liters/MWh of water requirements.
Stirling Dish (SD)				Cooling water not required. Around 19 liters/MWh for mirror cleaning and domestic use.

In liters/MWh.

There is considerable uncertainty in each of the above estimates given the limited experience with solar thermal power plants.

It is difficult to indicate the water use intensity of hybrid cooling. Such cooling systems may be designed to optimize the proportion of days with wet cooling on one hand, and the efficiency penalty associated with dry cooling on the other. The DOE report on which the above estimates are based (DOE 2009) estimates a function relating fraction of wet cooling tower water consumption (from 0% for dry cooling to 100% for wet recirculating cooling) with likely power penalty. Further, the penetration of these technologies is quite limited – currently the technology is not represented in California's power sector. Two approved Stirling Dish projects (Calico Solar or SES I and Imperial Valley Solar or SES II) were both withdrawn by the applicants recently due to "changed market conditions."

Table 14. Approved CSP Projects in California.

Project name	Solar Technology	Cooling type	Notes	Liters / MWh
Beacon Solar Energy (250 MW)	Parabolic Trough	Wet	Recycled water for cooling. Fresh groundwater for mirror washing and domestic use. ZLD system is a combination of osmosis and then evaporation ponds.	2,725
Abengoa Mojave (250 MW)	Parabolic Trough	Wet	Slightly brackish groundwater with TDS of 1,200 – 1,500 mg/L. ZLD system consists of evaporation ponds.	2,775
Blythe Solar (1000 MW)	Parabolic Trough	Dry	Groundwater with TDS ranging from 470 – 5,600 mg/L with high fluoride and chloride contents. ZLD system is a combination of osmosis and then evaporation ponds. Water used for auxiliary equipment cooling.	356
Genesis Solar (250 MW)	Parabolic Trough	Dry	Brackish groundwater with 3,000 - 5,000 mg/L salt concentration. ZLD.	409
Palen Solar (500 MW)	Parabolic Trough	Dry	As above	371
Rice (150 MW)	Tower (Tw)	Dry	Brackish groundwater with 660- 3,000 mg/L salt concentration – the project will largely tap water from wells with less than 1,000 mg/L. Disposal to evaporation ponds.	322
Ivanpah Solar (370 MW)	Tower	Dry		106

<u>Assumptions</u>

Per CA-TIMES electricity forecasts, there is limited growth in solar-thermal electricity production between now and 2030. It is assumed that the entire electricity demand in 2030 will be met by the current stock of power plants (see Table 14 above). Hence no assumptions are made about share of different cooling technologies, as well as sources of water.

E.3 Geothermal Power

Water requirements for geothermal electricity depends upon:

(a) *Resource enthalpy* – temperature of the inlet geothermal fluid. Resource enthalpy drives the type of power plant, energy conversion efficiency, and cooling water requirements.
(b) *Type of resource* – hydrothermal resource or enhanced geothermal systems.
(c) Cooling technology

Water is required for three purposes:

(a) Cooling and power plant water requirements;
(b) Water for heating and mining;
(c) Water injection into a geothermal reservoir to prevent decline in pressure and hence in production.

Each of the factors affecting power plant water requirements; and the different kinds of requirements, are discussed here, and the reader is referred to Mishra, Glassley et al. (2011) for a detailed discussion.

Resource Enthalpy

Low enthalpy resources. For geothermal resources with a wellhead temperature of less than 200 ◦ C, binary or Organic Rankine cycle (ORC) plants are most appropriate. These plants are very similar to Rankine cycle coal or nuclear plants, but use an organic fluid like iso-pentane instead of water as the working fluid. Energy conversion efficiency increases linearly with resource enthalpy; hence amount of waste heat that needs to be dissipated and consequent cooling water requirements (assuming wet cooling) decreases linearly.

Medium to high enthalpy resources. For geothermal resources with a wellhead temperature higher than 175–200 ◦ C, flash power plants are most appropriate. Most or nearly all the makeup water for cooling is provided by steam condensate. Additional water may be required during summer, reflecting the effect of higher ambient temperatures on water evaporation. Around 20-30% of the geothermal fluid extracted is lost through evaporation of the steam condensate in the cooling tower, and the remaining 70-80% is re-injected back (DeMeo and Galdo 1997, DiPippo 2008). In Mishra, Glassley et al. (2011) it was assumed that this lost water needs to be compensated to sustain the resource, although no evidence was found that existing flash plants are actually compensating for such losses. As a result, this study did not consider any compensation by flash power plants.

Steam-dominated resources. In the steam-dominated resources in Geysers, north of City of San Francisco, the steam condensate available for reinjection constitutes only 10-15% of the mass of dry stream withdrawn for power generation. Around 42 million liters of highly treated municipal effluents from city of Santa Rosa and the Lake County, California is injected daily to maintain steam pressure (Sanyal and Enedy 2011).

Of the 13 TWh of geothermal electricity produced in CA today (6% of total electricity), around 50% is generated by the dry steam resources in Geysers. It should be noted that such dry-steam resources are unique and any increase in geothermal electricity production in the U.S. is unlikely to be from such resources. Low enthalpy resources with binary power plants account for ~20% of total and medium-high enthalpy resources using flash plants contribute to the remaining 30%. Most of the growth in capacity in the U.S. over last one decade has been in binary plants using low enthalpy resources ((Matek 2013)

Type of Resource – Hydrothermal vs. EGS

Conventional hydrothermal resources have their own supply of geothermal fluid. Hydrothermal resources can be classified into dry steam (or vapor) dominated fields (Geysers) and water-dominated fields. Water dominated fields are further classified into hot water fields and wet steam fields.

EGS resources have high temperature but contain little or no geothermal fluid, and are not very permeable. To exploit such resources, hydraulic fracturing must create a permeable reservoir and fluid from the surface must be pumped through the fractures to extract heat from the rocks. A pair of wells is drilled into the rocks terminating several hundred feet apart. Fluid, which is usually water, is injected under high pressure through the injection well, which creates an artificial reservoir. The fluid then returns to the surface through the production well, and thus transfers the heat to the surface as steam or hot water.

Ideally, a closed loop is created whereby cold water is pumped down the injection well and returned to the surface through the production well after passing through the hot, artificially fractured formation. However, losses may occur due to permeation and leakage from the fracture system to the surrounding rocks. The extent of losses will depend upon site-specific conditions like permeability of rocks, depth of the reservoir, as well as age. Losses are also a function of injection pressure - Tester, Anderson et al. (2006) note that high-injection pressures extends the fractures and increases permeability which in turn increase flow losses. Murphy, Drake et al. (1984) indicate a loss of 5% of EGS reservoir circulation flow rate based on experience from the EGS resource at Fenton Hill, New Mexico where losses during start of operation were greater than 10% but decreased after that. Duchane (1996) indicated water losses in the order of 1-2% of EGS reservoir circulation flow rate. DeMeo and Galdo (1997) have calculated water consumption by a EGS resource based on water losses of 5% and 15%.

Fluid losses should be below 10% for long-term viability of the resource (Barbier 2002). This is not only true for the very large volumes of freshwater lost in that way, but also for the additional pumping power required for the makeup water.

Fluid losses in an EGS resource present a trade-off decision to planners. Higher efficiency of energy conversion may require a wet or hybrid cooling tower (see next section), but will require lower geothermal mass flow rates. Water required for cooling purposes may be compensated by lower fluid losses during heat mining. Another area of trade-off is between flow losses and electricity generated – higher injection pressures increase flow rates and hence electricity produced but also increases flow losses (Tester, Anderson et al. 2006). A 5% loss rate is assumed in all calculations. Nearly all geothermal electricity produced in the world today is from hydrothermal resources.

Cooling Technology

Power output for an air cooled geothermal plant can decrease by up to 50% from winter to summer (Michaelides and Ryder 1992, Kanoğlu and Çengel 1999). However, air-cooled thermoelectric plants suffer a much lower drop in performance in summer (Maulbetsch and DiFilippo 2006) usually around 10% or less. Diurnal fluctuations may also be significant; DiPippo (2004) reports that power output of the air cooled bottoming binary cycle in Brady, Nevada was 33% lower at 6 pm than at 6 am in the morning, based on observations.

On the other hand, a number of binary power plants in California (e.g. Mammoth, Casa Diablo) as well as Nevada have dry cooled systems given the paucity of water in regions with rich geothermal resources. A few power plants (like Amedee and Wineagle in California) have implement hybrid cooling system to mitigate the energy penalty during peak times. Overall, around 30% of total ORC capacity in the U.S. (~800 MW) is dry cooled (DiPippo 2009, Matek 2013).

Flash power plants usually do not have dry cooling because of water available through steam condensation. However, DiPippo (2009) notes that a few flash power plants in New Zealand, Japan and Philippines have dry or wet-dry hybrid cooling. The Puna power plant in Hawaii is also dry cooled; however all existing flash plants in California are wet cooled.

Table 15. Existing Binary (ORC) Power Plants in California.

		MW	**Approximate T_{inlet} (degrees Celsius)**	**Cooling**
Amedee	Lassen	2	110	Hybrid
Brawley North	Imperial	50	170	Wet
Gould	Imperial	10		Wet
Mammoth (Casa Diablo)		42	170	Dry
East Mesa		90		Wet
Heber	Imperial	~ 100	165	Wet
Wabuska		1.7	104	Wet
Ormesa	Imperial	~50	150	Wet
SIGC	Imperial	36	170	Wet
Wineagle	Lassen	1	110	Hybrid
TOTAL		**~375**		

Table 16. Existing Flash (Including Flash-Binary) Power Plants in California.

		MW	Approximate T_{inlet} (degrees Celsius)	Cooling
Coso (Navy)	Inyo	200	200-350	Wet
CalEnergy	Imperial	10		Wet
Del Ranch	Imperial	38		Wet
Elmore	Imperial	38		Wet
GEM, ORMAT	Imperial	10	165	Wet
John L. Featherstoneface	Imperial	50		Wet
Leathers	Imperial	39		Wet
Salton Sea	Imperial	150	250+	Wet
Vulcan	Imperial	35		Wet
Total		**~575**		

Assumptions

According to CA-TIMES, geothermal electricity output will grow from around 10.4 TWh today to around 14.5 TWh in 2030 in the carbon policy scenario; and 28.2 TWh in the *Reference* scenario. Some simplifying assumptions are taken regarding the resources that will contribute to this growth. Resource assessment done by USGS (Williams, Reed et al. 2008) for resources greater than 95 degrees Celsius and maximum depth of 6 km (Williams, Reed et al. 2008) indicates the following potential for geothermal electricity in California:

Table 17. Geothermal Power (MWe) and Electricity (Annual TWh) Potential in California.

	95% probability		50% probability	
	MWe	Annual TWh+	MWe	Annual TWh+
Identified Resources	2,422	19	5,140	40
Undiscovered Resources	3,256	26	9,532	75
EGS	32,300	254	47,100	371
Total	37,978	299	61,772	486

Note: + Assumes 90% capacity factor.

1. Share of Hydrothermal and EGS resources:
   - Hydrothermal resources are assumed to contribute to 80% of geothermal electricity in 2030 in California. EGS resources contribute the remaining 20%.
2. Share of resources based on enthalpy:
   - Electricity from dry steam resources is expected to remain at the current level (~6.5 TWh).

- Based on geothermal resource assessment by the MIT report ((Tester, Anderson et al. 2006), around 75% of the resources between 3 and 5 KM are at 150 degrees Celsius, and the remaining 25% are at 200 degrees or above. It is assumed that 75% of the remaining electricity is produced from low enthalpy resources using binary power plants; and remaining 25% use flash power plants.

3. Share of cooling technologies
   - In the *Baseline* water scenario, it is assumed that 35% of all binary power plants will have dry cooling – this is the U.S. average in 2010 based on DiPippo (2009) and Matek (2013). The remaining plants are water-cooled. In the *Smart* water-use scenario, dry, hybrid and wet cooling technologies have an equal market share. The Hybrid system is assumed to consume around 15% of the water of a recirculating water cooling system at an energy penalty of 4% (annual averages) (Mishra, Glassley et al. 2010).
4. Water sources:
   - For cooling purposes, both fresh and recycled water are assumed to have an equal market share in the *Baseline* water scenario. In the *Smart* water-use scenario, share of recycled water is assumed to be 75%.

E.4 Wind power

Consistent with Macknick, Newmark et al. (2012), it is assumed that wind electricity incurs no direct consumptive water use.

Appendix F. Water Use Impacts of Exploiting Oil Shale (Kerogen)

Shale oil is not to be confused with oil shale. The latter is defined as unconventional heavy oil (kerogen) found in sedimentary rock that contains solid bituminous materials and which is first heated, followed by the resultant liquid being captured, a process known as retorting. U.S. domestic oil shale reserves could total as much as three billion barrels, or roughly the same amount as the world's proven oil reserves (GAO 2011). Most of these reserves are part of the Green River Formation, located in parts of Colorado, Wyoming, and Utah. What follows is a review the sparse literature on oil shale mining, retorting, and refining technologies, as well as their associated water use impacts and intensity.

F.1 Technology Overview

A 2005 study estimates that the U.S. possesses vast reserves of potentially recoverable oil shale (here taking the restrictive definition that does not include tight oil) – the GAO (2011) estimates that roughly half of a total USGS estimate of 3 billion barrels of oil may be potentially recoverable. This is equivalent to the world's total proven reserves. Despite ongoing technological advances that might make exploitation economically feasible, domestic commercial development of these resources has not occurred, nor is the exploitable likely in the short- to mid-term.

The most comprehensive publicly available technical overview of the economics; engineering; and strategic, social, land-use and environmental impacts of shale oil development is a report written by the RAND corporation (Bartis, LaTourrette et al. 2005).

Bartis, LaTourrette et al. (2005) estimate that commercial scale oil shale operation would be capable of handling upwards of 50,000 barrels of oil per day, and as many as several hundred thousand barrels of oil per day. Mines designed to serve such plants will handle at least 25 million tons of output per year.

Two technologies have been tested at pilot scale for exploiting oil shale: (1) underground or surface mining following by surface retort, and (2) in-situ retorting where the oil shale is heated in place and the liquid extracted from the ground (Bartis 2005, 46). The first method (i.e. mining followed by surface retort) has been implemented at limited commercial scales outside the U.S. Germany, Brazil, China, and Estonia all operate small commercial plants.

After underground or surface level mining (production), shale oil is brought to the surface for retorting, a process in which it is heated to 480-540 degrees Celsius. This shale oil is then immediately upgraded ('stabilized') via hydrogen catalysis. Stabilized shale oil can be used as a feedstock for refineries and "should compete favorably with sweet, light crude oil" (Bartis, LaTourrette et al. 2005).

In-situ retort of shale oil requires heating the shale formation over a period of 2-3 years to temperatures of 340-370 degrees Celsius to separate the oil from the shale. A "freeze-wall" is created around the perimeter of the formation by pumping refrigerated fluid into the ground to prevent groundwater from flowing into the extraction zone. In 2005, Shell had conducted pilot-scale testing of these technologies in the Piceance Basin of northwestern Colorado, but the commercially viability and scalability of such a technique remains unproven (Bartis, LaTourrette et al. 2005).

The RAND report envisages a minimum period of twelve years for pilot project and RD&D development before technology learning enables a phase of production growth. Since eight years have passed since that projection, and given sustained interest in the intervening period, it seems safe to assumption that the technological capacities will be developed quite soon, if they have not already developed to adequate standards for economic development by the present.

F.2 Water Use – Intensity and Water Quality Impacts

There are no peer-reviewed published estimates of the WUI of shale oil extraction and retorting operations. The lack of commercial production of oil shale limits the available data on water consumption. Here estimates from the grey literature are reviewed, including an industry report (NOSA 2012), a report commissioned and published by the Government Accountability Office (GAO 2011), and a research report by the RAND corporation (Bartis, LaTourrette et al. 2005).

Water Quality Impacts

Leaching of salts and toxic chemicals were identified as a water quality concern for both mining and surface retorting operations and for in-situ retorting (Bartis, LaTourrette et al. 2005). While practices are available to prevent leaching, in both cases the capacity to prevent contamination after production operations conclude is called into question. In the case of mining and surface retorting, water quality impacts may stem from mine discharging across the production chain as well as from leachate containing high salt concentration and low concentrations of arsenic, selenium, and other toxic chemicals, from retorted oil shale. Leaching is seen as the most imminent threat to water quality, in particularly when contextualized by the substantial investment of Colorado's state and local water bureaus to control salinity levels. Further, there are substantial uncertainties concerning the potential changes in aquifer properties that might result from in-situ retorting, including leakage and subsequent contamination of water into the extraction zone.

RAND recommends that, if any commercial-scale shale oil development occurs, a comprehensive research program be established to monitor and model water quality impacts. They suggest a program consisting of mathematical/spatial and hydrologic modeling, laboratory testing, and field monitoring. Even with such a research program, they caution that a full understanding of the water impacts would only begin to emerge after 6-8 years of sustained analysis.

Water Consumption Volumes - WUI

In producing shale oil three processes require substantial water volumes (Bartis, LaTourrette et al. 2005):

- Dust control for materials extraction, crushing, and transport;
- Cooling and reclaiming spent shale, and;

- Upgrading raw shale.

The most simplistic of the reports reviewed is the industry flyer released by the National Oil Shale Association (NOSA 2012), which took as its point of departure an allegedly representative WUI of 1.7 barrels of water/barrel of oil produced for in-situ retorting, and 2 barrels of water/barrel of oil produced by surface retorting. More usefully, the report does catalogue the many determinants of WUI in shale oil operations:

- Whether upgrading is conducted on-site or remotely;
- Whether flushing is conducted for in-situ retorts, and whether this is done in an aquifer zone;
- Whether retorted oil shale requires water for stabilization;
- What power source (e.g. electricity, shale gas) is used for heating of formations in in-situ production and;
- How much hydrotreatment would be needed to produce syncrude or hydrofuels.

NOSA (2012) reports that water produced in retorting, amounting to 4-14% of shale oil produced, may be treated and then reused in production operations, thus reducing the water input requirements by 10%. The GAO confirms that recycling is unlikely to be economically viable (2011), and so is likely to be undertaken in the absence of regulations.

The GAO (2011) further explored the water impacts of oil shale deposits in three western states (Colorado, Utah, and Wyoming) that contain an estimated three billion barrels of oil, or about half of the world total proven reserves. Nearly three-quarters of the oil shale formation is beneath public (Bureau of Land Management) land, so the U.S. government will have considerable control over whether and how much of the reserve will be exploited. The GAO report focuses on four key issues:

- The current state of knowledge concerning surface- and groundwater impacts of oil shale operations, and federal research efforts to further investigate the impacts of oil shale exploitation;
- Volumes of water that would be required to develop the oil shale, and;
- Water availability in the regions in question.

Despite having funded research on the water impacts of oil shale development to the tune of roughly five billion dollars since 2006, the GAO report highlights the persistent uncertainty in our current state of knowledge concerning, *inter alia*, (a) not yet commercialized technologies, (b) potential commercially viable scale of eventual operations, (c) current water availability, and (d) characterization of water impacts.

Key findings regarding each issue are as follows:

Despite recent research conducted and commissioned by the DOE and the Interior Department, there was a consensus among officials and experts as to the gross inadequacy of data required to understand the "baseline conditions of water resources" for the study region. Further there was systematic lack of coordination in sharing research among federal agencies and among state water regulatory agencies.

Based on a review of available studies, the GAO estimates that the WUI for all oil shale production operations ranges from 1-12 barrels of water/barrel of oil produced from in-situ (underground heating) operations, and averaging about 5 barrels, and from 2-4 barrels of water/barrel of oil produced from surface retorting, and averaging about 3 barrels. It is worth noting that this estimate matches that reported in the 2005 RAND study (Bartis, LaTourrette et al. 2005), which cites other studies conducted in the early 1980s, with WUI estimates ranging from 2.1-5.2 barrels/barrel; and with a mean estimate of 3 barrels/barrel.

For the regions reviewed in the GAO report, adequate water is expected to be available for the initial phases of oil shale development, but competition from other end-users (e.g. municipal and industrial use) and for environmental needs (e.g. fish), as well as altered availability due to changing climate, are likely to lead to water availability eventually constraining development of the industry. The RAND study (Bartis, LaTourrette et al. 2005) cites water supply infrastructure (including pipelines, reservoirs, and groundwater development) as a likely constraint to eventual commercial operations in Colorado, and further notes that growing demand in municipal, industry, and energy sectors will likely compete for water supply with any infrastructure build-out by shale oil developers.

Table 18. Water Use Intensity by Activity and Water Quality (Functionality) of Oil Shale.

Process / activity	Min	Median	Max	Threshold quality / functionality	Detailed processes
Extraction / retorting	0 / 0.9	0.7 / 1.5	1.0 / 1.9	mostly recycled, some potable	Includes construction, transport (roads), dust control, mining, cooling of equipment and shale oil, steam production, & drinking & sanitation water needs
Upgrading / stabilization	0.6 / 0	0.9 / 0	1.6 / 0	recycled or freshwater	Process & cooling water requirements for processes (e.g. hydrodesulfurization & hydrodenitrogenation)
Power generation	0.1 / 0.6	1.5 / 0.3	3.4 / 0.9	recycled or freshwater	Assuming a range of feedstocks & cooling systems (e.g. coal-fired, NGCC, air, hybrid, & recirculating cooling)
Reclamation	0 / 0.6	1.4 / 0.7	5.5 / 0.8	recycled (pending technological & economic viability & regulatory allowance	To cool, compact, and stabilize waste shale rock, for revegetation of disturbed land and mines. For *in-situ* retorting, reclamation of subsurface zones may require considerable water to remove hydrocarbons
Domestic use	0.1 / 0.3	0.3 / 0.3	0.3 / 0.4	mostly potable	To support employees & their families in the region
Total	***0.8 / 1.8***	***4.8 / 2.8***	***11.8 / 4.0***		

Ratios are reported for in-situ / surface retorting. Water volumes are reported for in-situ/surface retort across the production lifecycle of oil shale, in barrels water/barrels retorted, upgraded oil shale produced.

Based on a review of six studies,[5] the GAO reports the range of water consumption estimates, broken down into five activities. Table 18 shows the minimum water source/functionality

[5] The studies are listed in Tables 8 and 9 of ***Appendix I*** of the 2010 GAO report: ENERGY-WATER NEXUS: A Better and Coordinated Understanding of Water Resources Could Help Mitigate the Impacts of Potential Oil Shale Development. Available at: http://www.gao.gov/assets/320/311896.pdf.

requirements of each process associated with oil shale production, retorting and upgrading. Note that the GAO reports barrels water 'used' per barrel retorted, upgraded oil shale, which trades with and has the properties of sweet crude for use in refineries. The original GAO estimates reported in Table 7.18 do not break up water use by quality / functionality, nor by whether the water is withdrawn or consumed.

Appendix G. Water Use Impacts of Developing the Monterey Shale

G.1 California's Regulations on Fracking and Developing the Monterey Shale

Currently, regulations and regulatory oversight of hydraulic fracturing is weak. Permits are not required for fracking, and Division of Oil and Gas Resources (DOGGR), the agency responsible for monitoring oil and gas development does not collect information on developments using fracturing techniques in the state, and so has no data on the number or intensity of fracking operations in California. Industry estimates that roughly one-third of oil production wells use hydraulic fracturing.

In April 2012 a bill passed the CA State Assembly's Natural Resource Committee requiring companies to disclose water volumes used in oil and gas (including fracking) operations. Then in June 2013, the State Assembly voted down Assembly Bill 1323 to ban 'fracking' within CA. This is the first and strongest of three bills to be put before the Assembly, with the second and third both mandating a bans dependent on a review of environmental impacts. These later bills will be voted on in 2014 at the earliest.

Nevertheless, fracking *in vertical wells* is already common practice and is growing in Bakersfield and elsewhere in the state. The California Independent Petroleum Association estimates that roughly one-third of oil and gas wells use hydraulic fracturing methods. Moreover, according to the New York Times, since 2010, advances in drilling technology and rising prices have ushered in increasing output in some regions of the Monterey deposits of more than 50% percent (Onishi 2013). Advances in a technique that has been used for decades – namely 'acid jobs,' or using hydrofluoric or hydrochloric acid to clean out well bores and to fracture rock containing oil and gas reserves, as well as other EOR and fracking methods, have made new areas outside of Bakersfield economical for development. But directional drilling methods in combination with fracking have not yet been applied to develop the deep tight oil deposits of the Monterey Shale.

Projections of Future Development & WUI Implications

A controversial 2013 study conducted by researchers at the University of Southern California (Hopkins 2013), funded by the Western States Petroleum Association, estimated the alleged economic benefits of horizontal fracking of the Monterey Shale. It concluded that development would bring 500,000 jobs, $24.6 billion in annual state and local tax revenue streams by 2020. Two key caveats should be emphasized in the study: first, one of two footnotes mention that the methodology is that of an 'impact study,' and not a cost-benefit analysis – where the former only measures a subset of the benefits, and leaves all costs unquantified. Second, the authors note that their production estimates should be considered both 'tentative' and 'very optimistic' as yet untapped wells may not be as productive as estimated.

Despite these methodological shortcomings, the median development scenario adopted by USC (Hopkins 2013) seems most accurate.