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Measuring and moderating the water resource impact of biofuel production and trade

By

Kevin Robert Fingerman

A dissertation submitted in partial satisfaction of the

requirements for the degree of

Doctor of Philosophy

in

Energy and Resources

in the

Graduate Division

of the

University of California, Berkeley

Committee in charge:

Professor Daniel M. Kammen, Co-chair
Associate Professor Margaret S. Torn, Co-chair
Professor Michael H. O'Hare
Professor David J. Vogel

Spring, 2012

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Abstract

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University of California, Berkeley

Professors Daniel Kammen and Margaret Torn, Co-Chairs

Energy systems and water resources are inextricably linked, especially in the case of bioenergy, which can require up to three orders of magnitude more water than other energy carriers. Water scarcity already affects about 1 in 5 people globally, and stands to be exacerbated in many locales by current biofuel expansion plans. This dissertation engages with several of the analytical and governance challenges raised by this connection between bioenergy expansion and global water resources.

My examination begins with an overview of important concepts in water resource analysis, followed by a review of current literature on the water impacts of most major energy pathways. I then report on a case study of ethanol fuel in California. This work employed a coupled agro-climatic and life cycle assessment (LCA) model to estimate the water resource impacts of several bioenergy expansion scenarios at a county-level resolution. It shows that ethanol production in California regularly consumes more than 1000 gallons of water per gallon of fuel produced, and that 99% of life-cycle water consumption occurs in the feedstock cultivation phase.

This analysis then delves into the complexity of life cycle impact assessment for water resources. Despite improvements in water accounting methods, impact assessment must contend with the fact that different water sources are not necessarily commensurable, and that impacts depend on the state of the resource base that is drawn upon. I adapt water footprinting and LCA techniques to the bioenergy context, describing comprehensive inventory approaches and developing a process for characterizing (weighting) consumption values to enable comparison across resource bases. This process draws on metrics of water stress, accounting for environmental flow requirements, climatic variability, and non-linearity of water stress effects.

My assessment framework was developed in hopes that it would be useful in managing the risks and impacts it describes. The primary actors in this governance effort are government regulators, whose policies and incentives continue to drive and to shape the expansion of the bioenergy industry. However, the ability of governments to manage the impacts of biofuels is severely constrained by their obligations under international trade law. This dissertation concludes, therefore, with

a detailed investigation into relevant precedents under the General Agreement on Tariffs and Trade (GATT) and the World Trade Organization (WTO). I use these precedents to identify the policy tools that governments would be able to bring to bear in moderating the water resource impacts and myriad other environmental and social concerns raised by bioenergy expansion.

"Global issues of trade, energy, and subsidies have an impact on water use, but water is rarely a main topic of global discussions and agreements on these topics"

David Molden, 2007

"When it comes to bioenergy, we have made the strategic decision to shift from foreign oil to domestic water"

Michael Webber, 2010

For Melissa and Nico

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ACKNOWLEDGMENTS:
First and foremost, I want to thank my family. Thanks go to Nico for inspiration, to Melissa for her love and partnership, and to my parents Dan & Lynne and sister Beth for all of their love and support over the years. My family, to which I am eternally grateful, extends far beyond this nucleus – to the community in which I was raised and the community I have built over the years. I couldn't have done this without them.

I also want to express my gratitude to my faculty advisers – Margaret Torn, Daniel Kammen, Michael O'Hare, and David Vogel – for aiding and advising me in countless ways throughout this process.

My work would not be what it is, and I would not be who I am, without the ERG community. In particular, Isha Ray and the Water Group provided a fertile ground for exploring ideas and developing intellectually. I am also grateful to Andy Jones and Sintana Vergara for their support, commiseration, and encouragement in our dissertation writing group.

I have had the opportunity to engage with some very interesting real-world problems over the past several years, as organizations have welcomed me into their decision-making processes. In particular, I am grateful to the members of the Roundtable on Sustainable Biofuels for electing me to their Steering Board and to my fellow board members for their commitment to our shared vision of an environmentally and socially responsible bioenergy industry. Specifically, I would like to thank Charlotte Opal, Martina Otto, Barbara Bramble, and Khoo Hock Aun for their effort and partnership.

I have benefited from the guidance of countless collaborators and advisers over these past many years. In particular, I owe a debt of gratitude to my co-authors, Göran Berndes, Pim Vugteveen, Brian Richter, and Stuart Orr, for helping me think through the ideas that comprise much of chapter 4 of this dissertation. I am also grateful for the research support, data, and feedback provided by Rich Plevin, Sonia Yeh, Morteza Orang, Scott Hayes, Andrew Guzman, and Daniel Harman.

Finally, I will always owe a debt of gratitude to Alex Farrell for teaching me by example how to be an effective, engaged, skeptical, and passionate scholar.

CHAPTER 1:

INTRODUCTION

1.1 Growing global demand for interconnected resources

Population growth and shifting demographics have led to a steady increase in global demand for water, food, land, and energy resources. Barring any major changes in consumption patterns, global agricultural production will have to grow 70% by 2050 (Bruinsma 2003). Even under optimistic assumptions regarding productivity and technology, this will entail at least a 10% increase in cultivated land and a 20% increase in agricultural water demand (De Fraiture, Wichelns et al. 2007).

At the same time, energy use is projected to increase even more quickly. The International Energy Agency has projected a 40% increase in total energy demand and over 55% increase in electricity demand by 2030 (IEA 2009). Some rapidly developing countries make up the vast majority of this increase. For example, almost 70% of anticipated petroleum demand growth through 2030 is projected to come from India and China (IEA 2009).

While some of this demand can be attributed to ongoing population growth, much of it is also due to changing global demographics. The middle class is growing rapidly in many regions, especially in Asia, where this demographic tripled in size between 1990 and 2008 (The Economist 2011). Many aspects of this trend are certainly positive, especially for those enjoying newfound prosperity, but it also brings with it increased strain on limited resources as more people's consumption patterns begin to mirror those in the global North.

One of the hallmarks of these rising incomes is a dietary shift away from staple grains toward a diet heavier in meat, dairy, vegetables, oils, and sugar (Rosegrant, Cai et al. 2002; Bruinsma 2003). Meat demand in particular is projected to increase globally between 70% and 155% by 2050 (Wirsenius 2003). Animal products are a particularly inefficient food source due to the energy lost as feed crops are converted into animal biomass. For this reason, animals currently consume approximately two thirds of all plant biomass appropriated for human use, while only providing about 15% of all calories consumed by people (Wirsenius 2003).

Water, food, land, and energy resources are deeply interconnected. For example, agricultural production today uses 37.7% of global land surface (World Bank 2012) and about 7000km^3 of water annually, or about 3000L per person per day (Postel 1998; Molden 2007). These values stand to increase by as much as 110% by 2050 (De Fraiture, Wichelns et al. 2007) – even more if current biofuel expansion plans are fully implemented (Hoff 2011). These links already create human development risk, as there is a great deal of overlap (Hoff 2011) between the 1.2 billion people without adequate access to water (Molden 2007), the nearly 1 billion who are undernourished (UNDP 2006), and the 1.5 billion who are without access to electricity (IEA 2009).

1.2 Bioenergy

Bioenergy is an excellent example of the interconnected nature of the resources described above. It typically requires much more land and water than other energy systems, and because it diverts land from other provisioning services, it can negatively impact food security. Modern bioenergy[1] (liquid biofuels and bioelectricity) currently plays a modest role in the global energy system. Only about 1.4% of global electricity and just over 2% of global liquid fuel is derived from biomass today[2] (Fulton 2004; US EIA 2012). However, both the proportion of biogenic sources in the energy mix and the absolute amount of bioenergy produced are growing rapidly. Figure 1-1 lays out the expansion in global production of liquid biofuels from 2000-2010, illustrating the rapid growth in this industry in recent years. This trend shows no sign of abating, as projections indicate that global consumption of biofuels could as much as double by 2030 (IEA 2009).

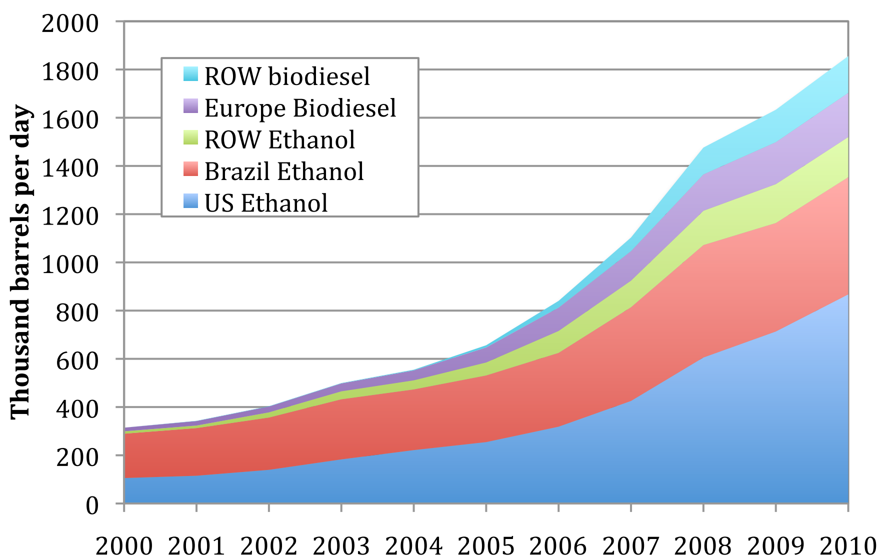

Figure 1-1: Growth in world biofuel production over the past decade

Prior to about 250 years ago, the vast majority of energy for human use came from biological sources, whether from the wood or charcoal used for heat and cooking or the crops that were fed to draft animals for work and transport. Since the

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[1] For our purposes, "bioenergy" is defined as energy derived from *recently living* organic material. Certainly, fossil energy sources such as petroleum, coal, and natural gas are also biogenic, in that they are the remnants of ancient biological material. However, energy from these fossil sources is not typically considered "bioenergy."

[2] These figures only include "modern" bioenergy. Traditional bioenergy (mainly wood, charcoal, and dung burned directly for heating and cooking needs) comprises a larger fraction of true global energy use. When these types of energy are included, it is estimated that about 7% of total human energy needs are supplied by biomass (Fulton, 2004).

decoupling of energy and biomass through the expansion of modern energy sources, total human energy consumption has risen rapidly to levels that make it difficult to return significantly to a biogenic energy system. To put the implications of any significant shift into perspective, total global energy demand today – about 500EJ per year – is an order of magnitude larger than the energy content of all food and feed crops produced annually (Gerten, Heinke et al. 2011).

Much of the recent past and projected future biofuel expansion is driven by policies aimed at increasing the share of renewable fuels in the energy sector. For example, in the United States, the current Renewable Fuel Standard (RFS2) mandates 15.2 billion gallons of various biofuels this year, rising to 36 billion gallons in 2022. In Europe, the Renewable Energy Directive (RED) sets an EU-wide target of 20% renewables in the overall energy mix and 10% in the liquid fuel mix by 2020. The RFS has been instrumental in driving the more than 3-fold increase in U.S. biofuel production since its inception in 2005 (US EIA2012). As a result of this expansion, about 40% of the U.S. corn crop is expected to be devoted to ethanol production this year (USDA 2012).

The global expansion in biofuel use, and its attendant increase in agricultural production and trade, raises an array of environmental and social concerns. Possible detrimental impacts range from greenhouse gas (GHG) emissions to biodiversity loss, labor abuses, reductions in food security, air pollution, and unsustainable use of water resources.

These concerns are all important. However, it is the last issue – the impact of biofuel production on water resources – that is the primary thrust of this dissertation. The water requirements of bioenergy systems range from 70 to about 400 times greater than those for fossil fuels and for other renewables such as wind and solar power (Gerbens-Leenes, Hoekstra et al. 2009). Because they are such a small part of the energy system, biofuels today exert very little pressure on water resources at a global level – accounting for only about 1% of all agricultural consumptive water use (De Fraiture, Giordano et al. 2008). However, the global water resource impact of these systems is expected to increase as bioenergy utilization continues to grow (De Fraiture and Berndes 2009).

On the local level, however, the water impacts of bioenergy production can already be much more severe. Biofuel and feedstock production in some important suppliers such as Canada, Russia, Brazil, and Indonesia are not meaningfully constrained by water availability. However, other producers such as China, India, South Africa, and to some extent the US and Argentina already face water scarcity issues, which would be exacerbated by further bioenergy expansion (Berndes 2008).

1.3 The Energy/Water nexus

Water and energy are inextricably linked. We use energy to move, treat, and heat water, and we use water for numerous activities in our energy systems. It is the latter of these linkages – the impacts of the energy sector on our water resources – that is primarily investigated in this dissertation. Water flows are used directly for hydroelectric power generation, but every type of energy production requires some degree of water use, whether for cooling, working fluid, refining, extraction, fabrication, cultivation, or any of a plethora of other activities involved in the life cycles of our electricity and liquid fuel systems.

1.3.1 Water for energy

Globally, about 8% of total water withdrawn is devoted to energy production systems. In some industrialized countries, this number climbs to 45% or more. As of the most recent United States Geological Survey report in 2005, thermoelectric power accounted for 49% of all water withdrawals in the country (Figure 1-2). These withdrawals have increased dramatically in recent years, in both absolute and relative terms. In 1950, thermoelectric power generation accounted for a scant 22% of withdrawals, and the total volume withdrawn was less than half of what it rose to by 2005 (Kenny, Barber et al. 2009).[3] This increase in use has brought us to the point where water scarcity is beginning to constrain energy infrastructure development (Gleick 2009).

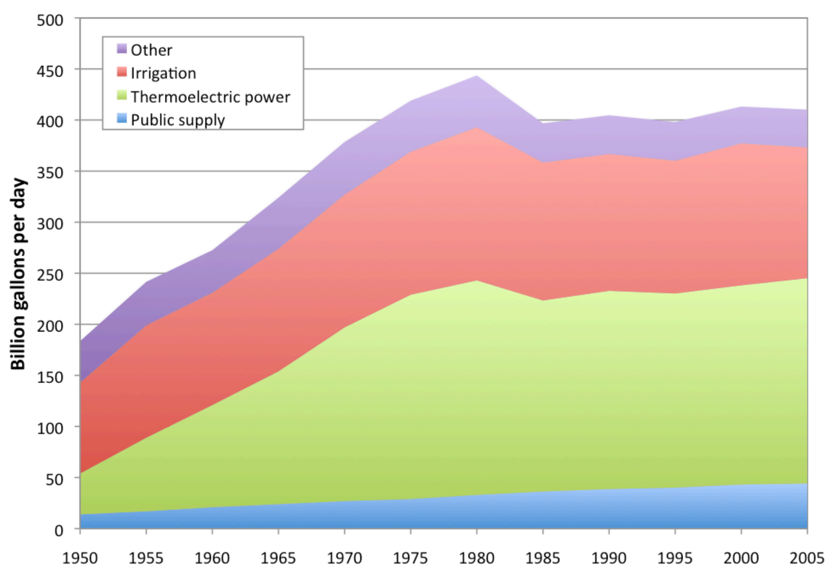

Figure 1-2: U.S. water withdrawals by sector 1950-2005. "Other" includes self-supplied domestic and industrial uses, as well as mining, livestock, and aquaculture. Data from Kenny *et al.*, 2009

[3] These statistics represent total water *withdrawn*. The largest *consumptive* use of water in the U.S. is agriculture. These terms and their relative importance are covered in detail later in this chapter.

Furthermore, some of the technologies that are coming to the fore as means to increase energy security are much more water intensive than current, average energy production. Domestic petroleum extracted from oil sands and shales or through secondary and tertiary recovery methods typically requires more water than conventional petroleum alternatives (Flint 2005). Similarly, bioenergy supply chains can be up to three orders of magnitude more water intensive than those required to produce petroleum fuels (King and Webber 2008; Gerbens-Leenes, Hoekstra et al. 2009; Fingerman, Torn et al. 2010).

1.3.2 Energy for water

Not only do energy systems consume water, but our modern water infrastructure also requires a great deal of energy for moving, heating, and treating water. These activities require so much energy that water systems are the dominant user of electricity in many municipalities (McMahon and Price 2011). Outside of cities, pumping of irrigation water is typically the primary water-related energy demand.

The energy intensity of water supplies can vary by over an order of magnitude depending on the source. For example, provision of a m^3 of clean, residential water from local surface flows requires about .37kWh while the same cubic meter of desalinated seawater requires 2.6-4.36 kWh (Webber 2008). Seawater desalination is an extremely energy intensive practice, so much so that new desalination infrastructure in the UK and Australia has recently been halted or delayed because of expected energy demands (Gleick 2009; Hoff 2011).

While the energy required for desalination is widely publicized, a much more ubiquitous process – that of simply *transporting* water – can also be hugely energy intensive. Bringing 1 m^3 of water over 350 horizontal kilometers requires an approximately equivalent amount of energy as desalinating the same volume (WBCSD 2009). Similarly, pumping of groundwater, the source of almost half of all irrigation water consumed, can be highly energy intensive, and only becomes more so as water tables continue to drop (Siebert, Burke et al. 2010). In some countries this use accounts for up to 40% of all energy consumption (Hoff 2011).

The connection between water and energy is particularly evident in California, where the State Water Project (SWP), carrying water over multiple mountain ranges to the state's arid South is the single largest consumer of energy. The SWP also, however, produces at its hydroelectric facilities more power than it consumes across its entire length (Cohen, Wolff et al. 2004). For agricultural water, transportation is the primary energy requirement. This is not, however, usually the case for residential water. Providing an acre-foot (326,000 gallons) of residential water to users in San Diego California requires a total of about 6900kWh. Of this, about 34% is used in transport while over 56% is consumed (mostly for heating), by the end users themselves[4] (Cohen, Wolff et al. 2004).

[4] The remaining 630KWh are used primarily for treatment of wastewater.

1.3.3 Interconnected risk

This interlinked nature between water resources and energy creates risks for both, since it means that strain on one of these resources often creates strain on the other.[5] The electric power sector, for example, is highly vulnerable to changes in water resource availability, including those expected to result from climate change (Vörösmarty, Green et al. 2000). Electric power plants are occasionally forced to shut down under drought conditions due to elevated discharge temperature or insufficient access to cooling water (Kimmell and Veil 2009). This is particularly challenging, as it is during these hot, dry periods that energy demand is at a peak. Similarly, water scarcity is a frequent source of conflict for the construction or continued operation of thermoelectric power facilities (Gleick 2009).

Because of these linkages between water and energy, consideration of the water resource effects of energy policy and production is both an environmental and a technical imperative. Peter Gleick of the Pacific Institute captured this issue in his 2009 remarks before the US Senate committee on energy and natural resources:

> *"Limits to the availability of both energy and water are beginning to affect the other, and these limits have direct implications for US economic and security interests. Yet energy and water issues are rarely integrated in policy. Considering them together offers substantial economic and environmental benefits (Gleick 2009)."*

There are times when a synergy exists between water conservation and energy policy goals. For example, reduction in oil sand extraction and refining due to its elevated life cycle GHG emission would also reduce the water consumption and pollution that often accompanies these activities. Moreover, the water sector offers policy options that can accomplish both preventative *and* adaptive goals in the face of global climate change. For example, water efficiency measures can help us adapt to increasing water scarcity in some regions, while at the same time reducing the GHG emissions associated with the current water infrastructure (Gleick 2009).

On the other hand, circumstances often exist in which the achievement of energy or climate goals and the preservation of water resources may be at odds. For example, concerns about indirect land use change in bioenergy systems could lead to irrigation of existing biomass crops, or to extensification of agriculture into uncultivated grasslands so as not to displace current production. These outcomes would mean net expansion in total water demand. Minimizing these conflicting incentives in water and energy, while at the same time taking advantage of the synergies between related goals will require truly integrated resource and policy planning.

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[5] The risks from this nested relationship were on acute display in the Fukushima Daiichi nuclear disaster in 2011, when a loss of electric power led to a loss of cooling water, which in turn led to the loss of large-scale electric infrastructure in a nuclear meltdown event.

1.4 Water scarcity

Water supplies worldwide are strained, with 1.8 billion people predicted to be living in absolute water scarcity and 2/3 of all people predicted to be experiencing some water stress by 2025 (UN Water 2007). This pressure is increasing rapidly, as population growth and dietary changes are projected to drive a 70-90% increase in demand for water worldwide in the next 50 years (Molden 2007). Climate change also stands to exacerbate water stress in many regions, intensifying desertification, reducing glacial storage, and increasing the frequency of extreme events such as droughts and floods (Hoff 2011). Furthermore, concerns about environmental and social impacts have curtailed new reservoir construction, while existing reservoirs are being lost to siltation and groundwater lost to overdraft at a rate of about 60-70 km^3 per year (Serageldin 2001), or about half the capacity of Egypt's Aswan High Dam reservoir.

About 70% of all water withdrawn annually by humans is used for agricultural purposes – up to more than 90% in some less developed countries (UNESCO 2009). This level of reliance on an increasingly scarce resource is beginning to constrain productivity in some globally important agricultural areas including California, South Asia, Mexico, Australia, and parts of China (Rosegrant, Cai et al. 2002; Shah, Burke et al. 2007; UNESCO 2009). Water resources are already being withdrawn to such an extent that that several important rivers, including the Yellow (China), the Syr Darya (Central Asia), the Colorado (Southwestern USA, Mexico), and the Murray-Darling (Australia), no longer reach the sea during some periods (Molle, Wester et al. 2007).

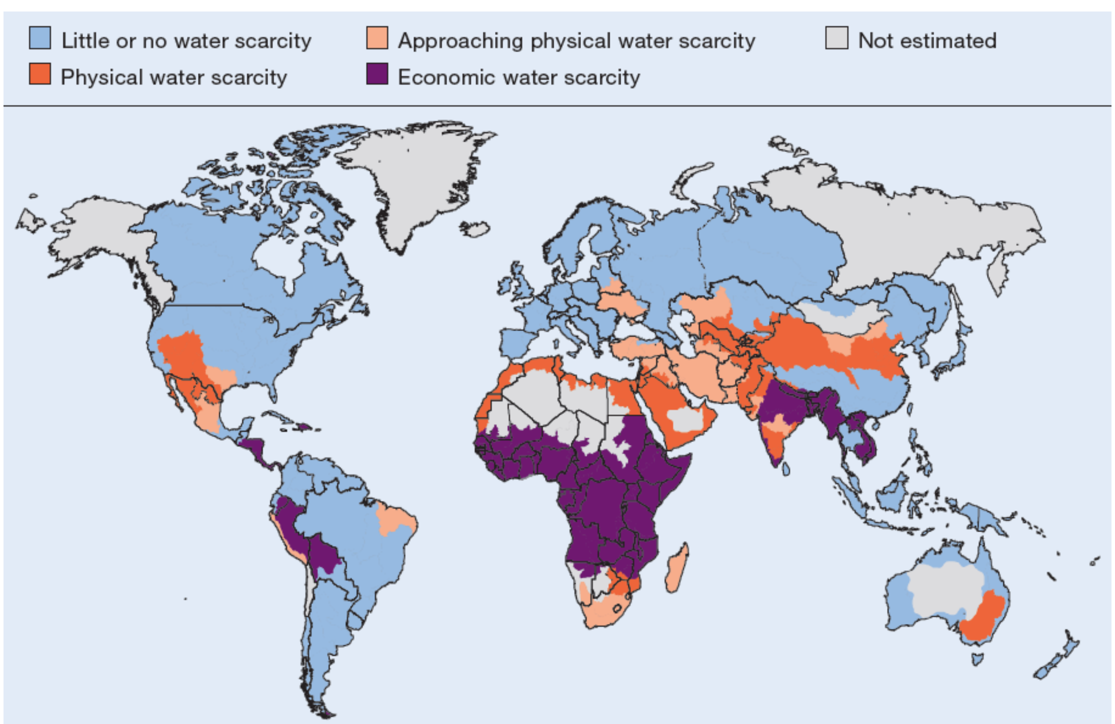

Figure 1-3: Geographic extent of physical (absolute volume) and economic (distribution infrastructure) water scarcity worldwide. (Molden 2007)

Beyond the direct constraints it places on human activity, water scarcity also has important effects on ecosystems. Notable impacts include aquatic and wetland habitat degradation, pollution effects, and soil salination (De Fraiture and Berndes 2009). This compromising of ecological integrity is also indirectly detrimental to human wellbeing through reduction of ecosystem services.

1.5 Types of water use

1.5.1 Withdrawal

Withdrawal is the removal of water from a natural system or a managed resource base. Irrigation is the primary driver of water withdrawal globally, using about 70% of all water withdrawn by humans. In the United States, irrigated area has expanded over five-fold in the last 100 years (USDA, 2009). In India, which relies to a large degree on groundwater for irrigation, its extraction increased almost 100-fold in the latter half of the 20th century (Hoff 2011). This rapid global expansion is due to the immense productivity improvements that can be gained through uptake of irrigation. While only about 15% of total cultivated land area is irrigated today, this area accounts for almost half of total crop production (Molden 2007).

Water is considered withdrawn regardless of its fate downstream. In the case of irrigation, all of the applied water is withdrawn, even though some of it infiltrates or runs off of the soil surface, later rejoining the usable resource base. Industrial cooling water is a more extreme example, as it is uses almost half of all water withdrawn in the United States, but returns over 95% of that water to the natural flow with little degradation beyond a small amount of waste heat (Kenny, Barber et al. 2009). Despite this, withdrawal is an important metric in many cases. For example, when groundwater is withdrawn, even if it is used only momentarily and then released to surface flows, it is lost to the original source. It is therefore the withdrawal rate that will determine whether current activities can be sustained or expanded.

The source from which water is drawn is also an important characteristic in assessing the impact of its use. For example, withdrawal of cooling or irrigation water from surface flows has very different implications than if it were drawn from groundwater sources, and both of these are very different than the withdrawal of a comparable volume of seawater.

1.5.2 Consumption

As discussed above, not all of the water that is withdrawn for use by humans is necessarily consumed in the process.[6] Furthermore, not all of the water that is

[6] Use of the term "consumption" is complicated by the fact that most of the processes being considered do not actually destroy water molecules. I rely here on a commonly used definition of water consumption; water is considered consumed when it is removed from the usable resource base for the remainder of one hydrologic cycle. Evaporation, therefore, is considered a form of consumption. Although the water has simply changed phases, we do

consumed by human activities has necessarily been applied. Agriculture is a good example of this; about 80% of all crop water requirement globally is met by rainfall (De Fraiture and Berndes 2009). Considering only irrigation implies that the 77% of US corn acreage that is exclusively rain-fed (USDA 2008) consumes no water. This is clearly not the case. If not devoted to biofuel feedstock production, rainwater can go to other productive uses – to cultivation of another crop, to environmental services, or to groundwater recharge (Molden 2007).

Cropping systems consume water in two ways: through evaporation from the soil surface and through transpiration, which is essentially the productive evaporation of water through plant tissues. These two processes are collectively referred to as evapotranspiration (ET). Table 1-1 lays out the application efficiency of various irrigation methods in California. Industrial processes consume water through evaporation in a broad array of activities, particularly cooling.

Table 1-1: Irrigation efficiency - as % of applied water consumed through crop ET. Adapted from Salas *et al* (2006)

Type of Irrigation System	Efficiency (%)
Surface Irrigation	
Furrow	67.5
Flood	60
Gravity	75
Sprinkler	
Hand move or portable	70
Center pivot/linear move	82.5
Solid set/permanent	75
Side roll sprinkler	70
Micro sprinkler	87.5
Trickle irrigation	
Surface drip	87.5
Buried drip	90

In accounting for different types of water consumption, researchers frequently make use of the concept of Green, Blue, and Grey water. **Green water** refers to rainwater and soil moisture that is naturally available *in situ* to the plant. **Blue water** is applied through human intervention – irrigation in the case of agriculture. **Grey water** is used to describe water consumed through pollution, and is discussed in more detail in the following section.

1.5.3 Pollution

Pollution is another important anthropogenic impact on water resources. It can be considered consumptive, in a way, since it removes water from productive use.

not control where evaporated water will fall next, so the water is functionally lost to the system.

It is difficult, however, to quantify water pollution in the same way that consumption and withdrawal are quantified, since there is no objectively correct method for defining how much water is "consumed" through the addition of a given mass of a pollutant. In approaching this issue, many researchers use the volume that would be required to dilute any pollutant discharge to below some acceptable level.

Nutrient pollution

Among energy technologies, nutrient pollution is of particular concern with biofuels, as they require an agricultural phase, with frequently large inputs of chemical fertilizers. Donner and Kucharik (2008) showed that the U.S. corn ethanol production targeted by the 2007 Energy Independence and Security Act (EISA) (U.S. Congress 2007) would result in a 10–34% increase in the export of dissolved inorganic nitrogen (DIN) from the Mississippi and Atchafalaya Rivers. Even without further expansion in biofuel, nutrient pollution in many waterways is acute. A recent US EPA report showed that approximately 32% of the nation's stream length shows high concentrations of nitrogen compared to reference conditions (Paulsen, Mayio et al. 2008)

Toxic chemical pollution

A variety of energy production methods can cause contamination of surface or groundwater with potentially toxic chemicals. Oilfields often produce large quantities of polluted water that can contaminate surface flows if released and so becomes a waste material that must be managed. Similarly, water used in removing the bitumen in tar sand petroleum production becomes mixed with the sand, residual oil, and process chemicals. Biofuel feedstock cultivation can also create toxic chemical pollution as agricultural chemicals such as pesticides and herbicides can find their way into ground and surface waters (USEPA, 2010).

Refinery processes can also result in environmental toxicity. Waste streams from petroleum refining can contain benzene, toluene, arsenic, and heavy metals.
In ethanol and petroleum refineries as well as thermoelectric power facilities, salt buildups in the cooling towers and brine byproduct from water purification create a waste stream that must both be periodically discharged (Berndes 2002; Keeney and Muller 2006; McMahon and Price 2011).

CHAPTER 2:

BACKGROUND ON ENERGY-WATER NEXUS

6.1 Introduction

The U.S. Energy Information Administration projects that total global energy[7] consumption will increase by 53% between 2008 and 2035, with electricity consumption rising by 82% over the same period (US EIA 2011). This continued growth has very important implications for water resources, and hence for ecology, agriculture, development, and geopolitics.

As total energy demand has risen and continues to rise, the relative shares provided by different energy technologies are also in flux. This fluctuation responds to economic and policy drivers, including decline or discovery of fossil resources, development of new technologies, and public policies such as those aimed at combating climate change. Transition to low-GHG energy sources, for example, can either increase *or* decrease total impact on water resource quality and quantity, depending on the technology pathways pursued. Among low-GHG energy sources, hydroelectricity and concentrating solar power consume more water than the existing grid mix. On the other hand, technologies such as solar photovoltaics and wind power require *very* little water to operate.

This chapter reviews some current research and data on the water resource impacts of most major energy production systems – information that is meant to provide context for the more detailed investigation of the water/biofuel nexus in subsequent chapters of this dissertation. The analysis takes as its system boundaries the ongoing operation – the *variable cost* in water – of each energy production process. While all of these technologies will consume some water upstream in, for example, the construction of electrical or refining infrastructure, this consumption is not considered here.

6.2 Biofuels

6.2.1 Purpose-grown feedstock cultivation

Some recent studies of biofuel water impact (King and Webber 2008; Chiu, Walseth et al. 2009; Service 2009; Wu, Mintz et al. 2009; Scown, Horvath et al. 2011) focus their analyses on water applied to agricultural fields as irrigation. When irrigation water is taken as the basis for calculating agricultural consumption, estimated life cycle water use for biofuels ranges from 10-324 L H_20/L fuel. The low end of this range represents the refining and transport consumption associated with biofuels made from unirrigated crops and waste materials (Service 2009).

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[7] These EIA projections look exclusively at “modern” energy carriers; they do not include some important energy sources such as biomass burned directly for heating and cooking.

Irrigation demand for a given crop can vary greatly depending upon climatic conditions and plant physiology. In Brazil, the world's largest producer of sugarcane, and second-largest producer of biofuel,[8] the crop is largely rainfed, with supplemental irrigation applied during critical periods in the Center-West region. In India, the worlds second-largest sugar producer, on the other hand, cane is grown mostly under full irrigation (De Fraiture and Berndes 2009).

In the U.S., corn in Illinois, Iowa, and Minnesota is generally rain-fed. However demand for biofuel has led to farmers' increasingly growing corn in more arid land farther west. This expansion is leading to increased pressure on already strained irrigation resources such as the vast Ogallala aquifer, which underlies eight western states and has dropped by more than 100 feet in some areas since it began to be significantly tapped for irrigation in the 1950s (McGuire 2009).

Irrigation water is a vital and unique resource, but rainwater is also of value. Many major biofuel crops are rainfed, including most Brazilian sugarcane and U.S. corn as well as much the majority of global oil palm, cassava, and rapeseed production (De Fraiture and Berndes 2009). This fact does not, however, mean that these crops consume no water. If not devoted to biofuel feedstock production, this green water could be allocated to other crops, to environmental services, or to reservoir and/or groundwater recharge (Molden 2007; Fingerman, Torn et al. 2010). For this reason, some studies have looked at all crop ET in an effort to comprehensively account for biofuel water consumption. This is typically done using some form of the Penman-Monteith model (Allen, Pereira et al. 1998), which estimates ET through a combination of crop physiology and climatic conditions such as solar radiation, wind speed, humidity, and temperature. Where crop ET is used to quantify agricultural water use, estimates of life-cycle water consumption for biofuels range from 380 to over 1500 L H_20/L EtOH (Dominguez-Faus, Powers et al. 2009; Fingerman, Kammen et al. 2009; Gerbens-Leenes, Hoekstra et al. 2009).

Berndes (2002) looked at the water impact of bioenergy expansion at a global scale, investigating both conventional (1^{st} generation) crops as well as biomass crops such as *miscanthus*. Similarly, Gerbens-Leenes *et al.* (2009) used an agro-climatic model to estimate global average water intensity of fuels from a variety of common biofuel feedstocks. Fingerman et al. (2010) expanded on this work, accounting for spatial heterogeneity stemming from differences in crop physiology, management, energy yield, and climate. In total, DeFraiture and Berndes (2009) estimate that 1.4% of global crop ET was devoted to biofuels. The estimated share of irrigation water was slightly higher (about 1.7%) due to the amount of irrigated (or partially irrigated) sugarcane in the biofuel mix.

[8] Brazil was the largest producer of biofuel in the world until it was overtaken in 2006 by the United States, which has subsequently continued increasing its output. As of 2010, the U.S. produced about 70% more biofuel than Brazil according to figures from the U.S. Energy Information Administration (2012).

6.2.2 Waste/residue feedstock collection

Some biofuel feedstocks can come from waste streams such as forestry litter, agricultural residues, municipal solid waste, used cooking oil, and others. In the case of agricultural wastes, their use as a biofuel feedstock can increase the total energy production from a cultivation system, thereby lowering the water intensity per unit output. For example, only 25% of aboveground sugarcane biomass is sugar and the grains only represent about 50% of aboveground biomass in cereals (Berndes 2008). The remaining removed biomass can be used to increase the output from cultivation systems without jeopardizing overall food, feed, and fiber production.

Materials such as forest wastes require no irrigation water, and no significant water is used in their collection, so these feedstocks are commonly assigned a water intensity of zero. The case of agricultural residues such as corn stover and sugarcane bagasse is more complex, in that irrigation water and polluting chemical inputs may be used in cultivation of the primary crop. There is some dispute over how to allocate impact between different products when multiple co-products (e.g. corn kernels and stover, or soil oil and meal) are the result of a single production activity. However, when the material used in biofuel production is a *true* waste product (i.e. would be disposed of or abandoned if not put to use as a biofuel feedstock) it is typically assigned none of the upstream water consumption or pollution.

6.2.3 Refining

Some studies of the water resource impacts of biofuel production only consider uses at the biorefinery (Keeney and Muller 2006; Schnoor 2007). This industrial phase requires much less water than feedstock cultivation, and so should not be the only consideration. However, because refining activity is spatially concentrated compared to feedstock production, this phase can have a significant local impact even when its share of the total life cycle water intensity is low.

First generation biofuels

For each 1 million gallons per year of production capacity, corn ethanol plants use enough water to support a town of approximately 5,000 people (Keeney and Muller 2006). An average corn ethanol plant consumes about 3.6 gal H_2O/gal ethanol produced, an improvement of about 20% over the past decade (Wu 2008). Dry mills use slightly less than this – as little as 3 gal H_2O/gal ethanol (Kwiatkowski, McAloon et al. 2006) – and wet mills slightly more, since they soak the grain prior to grinding, saccharification, and fermentation.

Biorefineries consume water through evaporation from cooling towers and boilers and in the process of drying distillers dry grains and solubles (DDGS). Some water is also consumed through "drift" (loss of liquid water to air flow through the cooling tower), and "blowdown" of accumulated salts from boilers and cooling systems. As shown in Figure 2-1, losses from the cooling tower and the DDGS dryer account for 95% of total plant water use. A small amount of water can be contained in the ethanol product, as well as in the co-product distillers' grains, which are often sold wet in order to cut down on energy use.

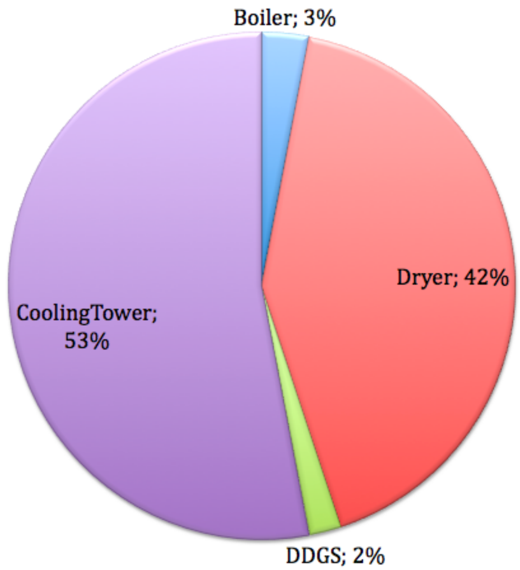

Figure 2-1: Water consumed in a corn dry mill ethanol refinery. Data from (Wu, Mintz et al. 2009)

In general, biodiesel production facilities use significantly less water than ethanol production facilities, as their feedstocks are hydrophobic oils, so no water is used in the conversion processes. Instead, water is used primarily to "wash" the finished product – removing any impurities such as remaining glycerin or incompletely reacted lipids. While water use in biodiesel production varies, it averages about 1 gallon per gallon of finished fuel (Pate, Hightower et al. 2007).

Biomass/cellulosic biofuels

Biochemical conversion of cellulosic biomass currently requires about 9.5 gal H_2O/gal EtOH (Schnoor 2007), though this figure could be reduced to about 5.9 gal/gal through relatively easy to achieve increases in yield (Aden, Ruth et al. 2002). By contrast, thermochemical conversion is more water efficient, consuming 2.3 gal/gal for pyrolysis of woody biomass (Jones, Valkenburg et al. 2009) and only 1.9 gal/gal for gasification and catalytic synthesis (Phillips, Aden et al. 2007).

Cellulosic feedstock material is washed prior to shredding in a system similar to that used for sugarcane. Used wash water is then recycled after being cleaned of solids (mostly dirt and fine biomass particles); the solids are then dewatered in a press and returned to the field. In general, about 1% of the wash water evaporates in the washing process, 3% is absorbed by the biomass, and less than 1% leaves the system with the soil and fine particulates (Aden, Ruth et al. 2002).

Water is also consumed in evaporative losses from boilers and cooling systems as well as contained in waste biomass streams that are removed from the facility or burned for process heat and electricity. Some of the chemical conversions involved in cellulosic fuel production technologies consume water. An example of this is the prehydrolysis process, which converts long chain polymers to fermentable sugars, splitting water molecules in the process.

6.2.4 Pollution effects

Feedstock

Several water quality and pollution impacts can result from biofuel feedstock cultivation and collection. Due to agrichemical inputs and tillage practices, cultivation commonly leads elevation in nutrient and other chemical levels in waterways. The U.S. Environmental Protection Agency expects increases in corn production for ethanol, for example, to lead to increases in the occurrence and concentration of nitrate, nitrite, atrazine and other contaminants in drinking water (US EPA, 2010).

Nutrient loading from fertilizer use leads to eutrophication and other ecological impacts. Corn, the primary biofuel feedstock in the United States, is a particularly polluting crop. It is much more heavily fertilized than the soy, cotton, or pasture it typically replaces, and only utilizes between 40% and 60% of applied nitrogen, leaving the rest to run off into the surrounding watershed or to form gaseous N emissions (US EPA, 2010). The Mississippi-Atchafalaya River Basin produces 80% of the nation's corn, and drains into the Gulf of Mexico, where it contributes to a "dead zone" larger than the state of Connecticut. This condition is expected to be further exacerbated by an estimated 10-34% increase in nutrient export resulting from federal biofuel mandates (Donner and Kucharik 2008).

Major factors affecting agricultural nutrient export to surface water flows include soil type, proximity to water bodies, tillage practices, crop rotation, and the use of tile drainage (US EPA, 2010). Feedstock choice will also strongly affect the water quality impacts of biofuel production. For example, biodiesel feedstocks, such as soybeans, as well as cellulosic feedstocks such as miscanthus, switchgrass, and poplar, are expected to have much lower fertilizer application rates than corn, significantly lowering projected water quality impacts (US EPA, 2010). Table 2-1 lays out projected rates of runoff.

Table 2-1: Chemical fertilizer runoff from cultivation of biofuel feedstocks in the United States. Fertilizer inputs per T of feedstock production were calculated based on data from GREET, National Agricultural Statistics Service (NASS), and FAOSTAT. Fractional runoff coefficients were derived from Johnes (1996).

		Corn (T)	Soybean (T)	Sugarcane (T)	Palm Oil (T)	Herbaceous Biomass (T)	Farmed Trees (T)
N	*(g.)*	2027.4	838.5	131	581.0	1406.8	93.8
P	*(g. as P_2O_5)*	74.8	208.6	2.9	36.7	3.8	5
K	(g. as K_2O)	204.7	692.9	6	91.7	6	8.8

Erosion presents another major cause of degradation in U.S. water resources, fouling wildlife habitat as well as carrying more persistent chemicals into the water, attached to soil particles. According to the National Research Council, cropland erosion is responsible for half of the sediment that is flushed into U.S. waterways annually, with more intensive cultivation causing greater erosion (Schnoor 2007). Use of waste and residue feedstocks reduces the fertilizer loading associated with purpose-grown biofuels. However, care must be taken to ensure that these materials are not removed from the field in sufficient quantity to impair soil structure and increase topsoil erosion.

Refining
Conversion of feedstock into biofuel can also have important water quality impacts. The groundwater that is available to most US biofuel production facilities is high in dissolved materials such as sulfate, iron, sodium, and carbonates (US EPA, 2010). These are generally present in too high a concentration for the water to be directly used in processing, leading facilities to purify their process water through reverse osmosis. This leaves a brine effluent that cannot be cleaned and must be disposed of. A similar brine results from the blowdown of deposits left on the surfaces of cooling towers and boilers by the ongoing evaporation of mineral-laden water.

Wash water from biodiesel post-processing is also an important potential source of water pollution as it contains nutrients and glycerin and can have a very high biological oxygen demand (GAO, 2009). Some producers recycle this water, though it must be purified, creating a further concentrated waste stream. As point sources of pollution, biofuel production facilities are regulated under the Clean Water Act, and must maintain a permit for any discharges.

6.2.5 Improving biofuel water impacts

For cultivated feedstock crops – whether conventional or second-generation – the primary task in mitigating water impacts is to alter the ratio between productive transpiration and non-productive evaporation in total crop ET. If this ratio can be shifted in favor of transpiration, productivity increases can be realized without increasing total consumptive use or affecting local water resource availability.[9] Cover cropping, intercropping, and improved soil and land management techniques can improve this balance (Berndes 2008).

Somewhat counterintuitively, the use of supplemental irrigation on previously unirrigated systems can greatly increase the overall water efficiency of bioenergy feedstock production. The application of new irrigation water will increase the total water use in the system, but significant yield improvements can reduce water use per unit energy produced. Supplemental irrigation may be cost prohibitive in some contexts, though policies directed at reducing indirect land use change could create a premium on the “additional” biomass generated through these yield increases. This would, in turn, make supplemental irrigation systems more economically attractive.

[9] This zero-cost dynamic does not apply when total ET is increased, however. Where surface flows are diverted for crop use, or where cultivation systems make better use of available rainwater, there can be a decrease in downstream surface and groundwater availability.

Use of true waste materials, such as municipal waste streams, some forestry residues, or some used oils, as bioenergy feedstocks can greatly improve the water efficiency of energy production, as these feedstocks are typically considered to contain no embedded water. In agricultural systems, crop residues and co-products can also be utilized for bioenergy production, further reducing water impacts per unit energy produced.[10] Furthermore, if residues and co-products of crop-based biofuels are put to use in non-energy systems such as animal feeds, this results in an indirect water savings through displacement of other material use (Berndes 2008). This displacement can be considered to offset some of the water used in cultivation in much the same way such calculations are performed for consequential life cycle assessment (LCA).[11]

Water quality impacts of bioenergy feedstock production can also be greatly reduced through better agricultural management. Cost-effective practices on cultivated lands include crop rotation, use of managed riparian buffer zones, appropriate timing and rate of nutrient application, use of treatment wetlands, and drainage management (US EPA 2010). Where crop residues are used as a feedstock, soil and nutrient runoff can be managed through carefully considered rate, timing, and method of biomass removal.

Biorefineries can also improve their water efficiency, through vapor capture, heat recycling, and other process optimization measures. Some in the ethanol industry assert that current technology could, with sufficient investment, allow for an ethanol refinery with zero net water consumption (Wu 2008). Through use of sugarcane bagasse, corn stover, or other biomass residues for cogeneration of process heat and power, other electric power production, and its attendant water consumption, is displaced. Also, where excess electricity can be exported to the grid, the total energy output from the system per unit of crop evapotranspiration increases.

Finally, a great deal of potential for water impact mitigation can also be found in the use of multi-functional biomass systems where bioenergy feedstock is produced by a system designed to provide synergistic environmental services. For example, some cultivated biomass can improve water quality, acting as vegetation filters for nutrient-rich waste streams from wastewater, landfill leachate, or agricultural runoff. Biomass-producing systems can also reduce sediment loading to waterways

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[10] It should be noted that this approach does not reduce the total water impact of cultivation. In fact, if not managed properly, residue removal can increase crop impact. However, it does increase productivity per cultivated land area, thereby potentially increasing water productivity and reducing impact per unit energy produced.

[11] Water use for goods displaced by bioenergy co-products would not necessarily have come from the same resource base as the bioenergy feedstock. Unlike well-mixed greenhouse gases, which have an effectively uniform impact wherever they are emitted, the impact of water consumption varies depending upon the state of the resource base being drawn upon. Chapter 4 of this dissertation lays out a methodology for managing this issue.

by reducing erosion through anchoring soil, improving infiltration, improving soil structure, and reducing the speed and volume and surface runoff (Berndes 2008). Finally, biomass plantations have the potential to reduce water quality risks by creating opportunities for phytoremediation of soil salinity or heavy metal contamination.

This integrated, multi-functional approach to bioenergy system design can also extend to the industrial processing phase of biofuel production, with beneficial outcomes for water resources. Careful siting and design of biorefineries will minimize conflicts between different water uses. Co-location with wastewater treatment facilities allows biorefineries to make use of degraded effluents for many process needs. Co-location with livestock operations allows for cycling of water and waste products between the two processes, including the efficient use of wet distiller's grains as cattle feed.

6.3 Petroleum fuels

As of 2009, petroleum fuels provided about 35% of global primary energy use and about 40% of primary energy used in the United States. Domestic production accounts for about 55% of US petroleum use, a figure that has generally declined over the past 30 years except for periods of increase during the 1970s energy crisis and the financial crisis of the past several years (US EIA 2012). For petroleum imports in 2011, Canada, Mexico, and Saudi Arabia were the three largest suppliers, accounting for 23.8%, 10.6%, and 10.5% respectively, followed by Russia, Nigeria, and Venezuela (US EIA 2012).

Production of petroleum fuels requires water in both extraction and refining. The amount of water used in petroleum production varies greatly, mostly due to differences in the source and type of crude oil and the extraction technology being employed. Water pollution effects can also be an important cause of concern throughout petroleum life cycles from exploration to end-use. Waste streams and leachates from petroleum extraction and processing can contain toxics such as benzene, toluene, and xylene, arsenic, and other heavy metals, as well as various organic compounds and residual hydrocarbons (Wu, Mintz et al. 2009).

6.3.1 Conventional petroleum extraction

From a water resource standpoint, petroleum extraction can be largely divided into three basic types:

- *Primary recovery* is the type of extraction that is familiar to most lay people. During primary recovery, pressure in the underground oil-bearing formation is sufficient to drive the oil to the surface. This can be due to a number of factors including natural water displacement, expanding natural gas reservoirs, and gravity drainage.

- *Secondary recovery* is necessary once the underground pressure can no longer drive the oil to the surface. At this point, the oil-bearing formation is flooded with a fluid, usually water, to force more oil to the surface.
- *Enhanced Oil Recovery (EOR)* is used once additional pressure in the formation is no longer sufficient to drive the oil to the surface because it is trapped by surface tension or viscosity. The methods for EOR vary greatly. In some cases CO_2 is injected into the well to reduce the surface tension of oil that remains. In other cases, steam or very hot air are injected in order to heat the formations, decreasing the viscosity of the petroleum, and making it easier to bring to the surface.

Each of these extraction methods requires some water, generally for injection into the oil-bearing formation. Table 2-2 outlines the average total injection water consumed for each extraction technique. Water that is present in the oil-bearing formations – whether naturally or injected as part of the extraction process – is sometimes pumped to the surface along with the crude oil. This "produced water" is the single largest waste stream created by the oil and gas industries (API 2000). This water can contain some of the chemical characteristics of the petroleum itself, including dissolved salts, organic chemical compounds, and naturally occurring radioactive material (Veil, Puder et al. 2004).

Table 2-2: Injection water required for production of crude oil through primary, secondary, and various enhanced recovery methods. Adapted from (Wu, Mintz et al. 2009). Some of this injection requirement will be made up from produced water extracted with the oil, reducing both the requirement for new freshwater resources and the pollution effect from disposal of produced water.

Production method	Gal H_2O /gal crude	% of U.S. crude production
Primary Recovery$^{\alpha}$	0.2	6.6%
Secondary (Water Flooding)$^{\beta}$	8.6	74.7%
EOR (Steam injection)$^{\alpha}$	5.4	8.2%
EOR (CO2 Injection)$^{\chi}$	13.0	6.8%
EOR (forward combustion)$^{\alpha}$	1.9	0.4%
Other EOR$^{\delta}$	8.7	3.2%
U.S. Average	8.1	100%

$^{\alpha}$ (Gleick 1994)
$^{\beta}$ (Bush et al, 1968)
$^{\chi}$ (Royce, Kaplan et al. 1984)
$^{\delta}$ (Wu, Mintz et al. 2009)

In many cases, produced water can be re-injected into formations as part of ongoing extraction activities. This approach can offset some or all of the water use required for the extraction method as outlined in Table 2-2. Wu et al*.* (2009) use data from

Veil et al. (2004) to estimate the US average produced water to oil ratio at 6.8 gallons produced water for every gallon of crude extracted. According to a survey by the American Petroleum Institute (2001), 71% of this produced water is re-injected nationwide. Using these figures, along with the U.S. production-weighted average reported in Table 2-2, the estimated average *net* water consumption for production of crude oil in the U.S. is 3.2 gal H_2O/gal crude.

Saudi Arabia is the world's largest oil producer, though very little information is available as to its production practices. Oil production consumes more water than any other activity in Saudi Arabia, and groundwater overdraft is a perennial problem. Operations at Ghawar, the largest oilfield in the world, use about 7 million gallons of treated seawater per day in the production of about 5 million barrels of crude oil (McMahon and Price 2011). Elsewhere in Saudi Arabia, oil production requires injection of 3 or more gallons of treated seawater for every gallon of crude produced (Wu, Mintz et al. 2009).

6.3.2 Oil sands petroleum extraction

Oil sand refers to a mixture of hydrocarbons called bitumen that is present in a geological formation of sand or porous rock. Interest in oil sand is currently very high, as reserves are large and are present in regions that are often more stable politically than some conventional petroleum sources. The majority of proven and exploited oil sand reserves exist in Canada, where production of crude oil has more than doubled since 2001, to almost 1.5 million barrels per day in 2010. This trend is expected to continue, with production rising to 3.7 million barrels per day by 2025 (CAPP 2011).

Canada's 179 billion barrels of proven reserves – 98% of it in oil sands (McMahon and Price 2011) – makes it the third most oil-rich nation on the planet after Saudi Arabia and Venezuela (Radler 2008). As of 2006, Alberta's oil sand mining operations were licensed to divert 370 million m^3 of water from the Athabasca River annually and planned operations would increase that figure to 579 million m^3. According to the National Energy Board (2006), "Stakeholders agree that the Athabasca River does not have sufficient flows to support the needs of all planned oil sands mining operations."

Because the viscous, tar-like bitumen is bound up in sandy deposits, it is more difficult than conventional petroleum to obtain in usable form. Three primary production techniques make up the vast majority of oil sand petroleum production. Their application depends on the location, type, and geomorphology of the oil-bearing deposits.

- *Mining* occurs when the oil sand deposit is less than 250 ft from the land surface. The oil-bearing feed ore is mined out of open pits, and the bitumen is removed from the matrix material after it is brought to the surface. Only 18% of known oil sand deposits can be extracted in this way, but they are very productive, as approximately 90% of the oil is recovered (Flint 2005). Where oil-bearing structures lay deeper underground, mining is more costly, so the bitumen must be removed from the deposit *in situ*. The two common *in situ* extraction methods are Cyclic Steam Stimulation (CSS) and Steam Assisted Gravity Drainage (SAGD).

- *Cyclic Steam Stimulation (CSS)* is used in deep and thick reserves with good horizontal permeability. Steam is injected into the formation, heating the bitumen to reduce its viscosity. The well is then shut to allow the formation to "soak" as the heat moves into the bitumen, causing it to flow more easily. Finally, the same well is then used to pump the bitumen out of the ground. CSS generally allows 20-35% of bitumen to be recovered from a formation (Flint 2005). Figure 2-2 presents a schematic of the CSS production method.

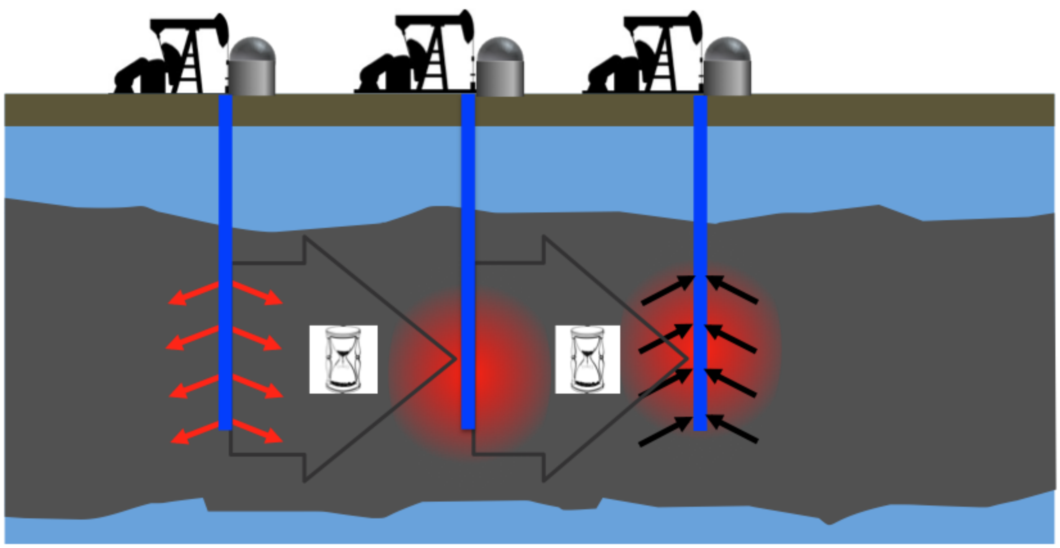

Figure 2-2: Cyclic Steam Stimulation (CSS) involves first injecting steam into a formation in order to heat the bitumen, thereby reducing its viscosity and making it easier to pump to the surface through the well shaft.

- *Steam Assisted Gravity Drainage (SAGD)* operates in a similar fashion to CSS in that it also uses hot steam to liquefy the bitumen. SAGD is deployed in comparatively thin and broad formations with good vertical permeability. Wells are drilled horizontally through the formation, with the injection well lying above the production well. Steam is then forced through the upper well bore, heating the upper portion of the oil-bearing formation. This causes the bitumen in this area to liquefy and then flow down to the production well aided by gravity. SAGD is capable of extracting 60-65% of bitumen from a formation (Flint 2005). Figure 2-3 presents a schematic of the SAGD bitumen production method.

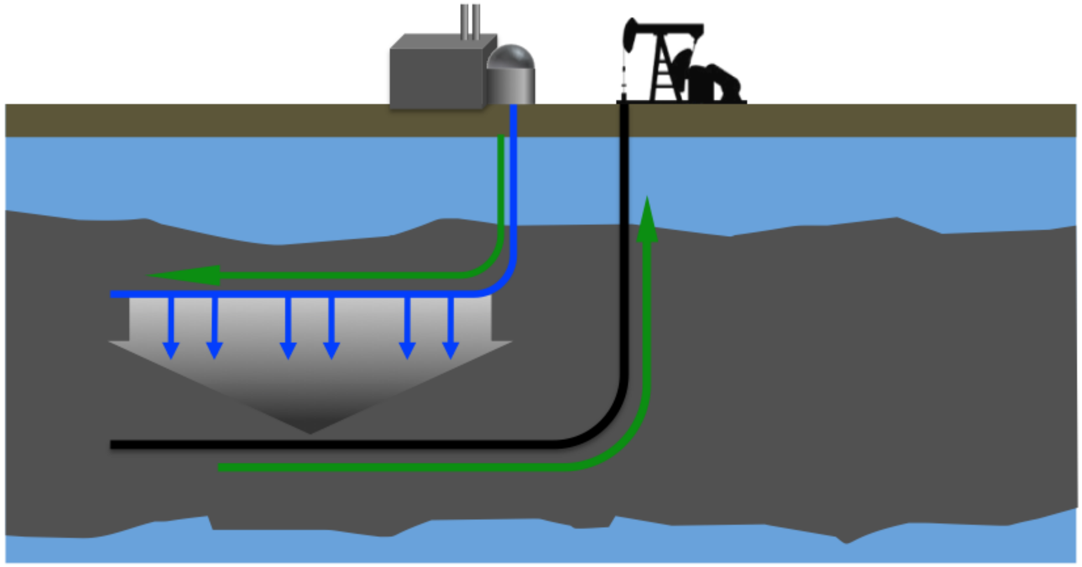

Figure 2-3: Steam Assisted Gravity Drainage (SAGD) involved injection of steam into the upper bore, heating the bitumen to make it flow more easily so that gravity carries it into the lower bore, through which it is extracted.

Surface mining of oil sand requires an extraction plant wherein the feed ore, mined from the oil-bearing deposit is heated using hot water and steam to separate the bitumen from the sand matrix. Solvents are then used to extract the bitumen from the bitumen froth (bitumen, along with water and fine sandy material) that results from the separation phase (Flint 2005). The remaining slurry of solid material and contaminated water are then sent to a tailing pond. Roughly two tons of oil sand are required to produce one barrel of crude oil (McMahon and Price 2011).

Bitumen extracted from oil sand deposits, whether through mining or through *in situ* technologies, must be upgraded into synthetic crude before it can be sent to a petroleum refinery. This process can take place where it is removed from the ground (as is usually the case with mining extraction) or can be done at a refinery if bitumen-ready pipelines and refineries are available. The upgrading process consumes about 1 gallon of water per gallon of crude (Wu, Mintz et al. 2009).

Surface mining requires approximately 4 gallons of water per gallon of synthetic crude (Peachey and Eng 2005). The mining itself is not very water intensive, but the processing of the feed ore requires a great deal of both steam and liquid water, which then ends up in tailing ponds from which it evaporates. *In situ* production is less water intensive since the bitumen is removed from the sands underground. SAGD requires less steam and so is generally less water intensive than CSS, though this may be as much a function of the geomorphologies of the formations into which these technologies are deployed rather than anything inherent to the technologies themselves (Wu, Mintz et al. 2009). Table 2-3 lays out the total water consumption for the three major methods of extraction and upgrading of bitumen from oil sand deposits.

Table 2-3: Net water consumption for major oil sand production and processing practices. All consumption is in gallons water per gallon of crude produced. Adapted from (Peachey and Eng 2005; Wu, Mintz et al. 2009)

Method	Extraction	Upgrading	Refining	Total	% share of production
Surface mining	4.0	-	1.5	5.5	56%
***In situ* - SAGD**	0.3	1.0	1.5	2.8	22%
In situ* - *CSS	1.2	1.0	1.5	3.7	22%
Weighted Average	2.6	0.44	1.5	4.5	100%

Surface mining is cheaper and removes more of the available bitumen from the formation than do *in situ* technologies, but it also has a much larger environmental impact due to its larger physical footprint as well as the tailings that it generates. *In situ* techniques produce less waste, because the sand remains underground, but still generate wastewater from produced water coming out of the wells as well as from the upgrading process. A zero dumping policy is typically maintained for these wastes due to their acute toxicity from organic acids and trace metals that leach out of the bitumen during extraction and processing (MacKinnon and Sethi 1993). As a result, almost all wastewater coming out of oil sand operations ends up in tailing ponds (Canadian National Energy Board 2006) which have grown to over 70 km^2 in total area and 700 million m^3 in volume (Allen 2008).

6.3.3 Refining

Upon arrival at the refinery, both conventional crude oil and synthetic crude produced from oil sand bitumen are refined into a variety of petroleum products. Estimates of water use in petroleum refineries range from 1-1.85 gal H_2O/gal gasoline (Gleick 1994). As with the biorefineries discussed above, most of this water is consumed through evaporation from cooling towers. The remainder is lost through drift (wind removal of liquid water from cooling towers) as well as blowdown (use of water for removal of mineral salt buildup on cooling towers) By 2025, 40% of global petroleum refining capacity is expected to be located in water-scarce regions (Wu, Mintz et al. 2009).

6.4 Electric power

6.4.1 Fossil energy extraction

1.09 billion tons of coal were mined in the United States in 2010, more than twice what was extracted fifty years earlier. Coal mining can occur on the surface or deep underground, depending on the depth of the resource.[12] Underground mining requires significant volumes of water for cooling and lubrication of cutting equipment as well as for dust suppression (Yergin and Frei 2009). The produced coal is also typically water processed to increase heating value and to reduce sulfur content. In the United States, coal mining and washing are estimated to use between

[12] Underground mining is typically employed for coal seams greater than 100 feet below the surface.

70 and 260 million gallons of water per day (Bian, Inyang et al. 2010). Coal mining is also having a significant impact on water resources in China, where production of coal has tripled since 2000. About one-fifth of China's annual freshwater consumption – or 120 billion m^3 – is used in coal production and consumption (Schneider, Turner et al. 2011).

Surface extraction is less water intensive than underground mining, primarily requiring water for reclamation and revegetation of mine sites after extraction is halted (DEO-NETL, 2006; McMahon and Price 2011). In recent decades, however, coal mining in the United States has moved towards surface extraction because of the shift away from the traditional mining regions of the East towards shallower, western coal deposits. These western coals contain less sulfur than their eastern counterparts, and so their production has expanded since the advent of sulfur dioxide (SO_2) emissions restrictions with the Clean Air Act Amendments of 1990.[13] While western coal requires less water to extract, it also has a lower heating value than eastern coal, leading to an increase in total coal extracted per unit of energy produced (DOE-NETL 2006).

The water required for coal production varies depending on factors including location of the mine, physical properties of the coal, and methods employed in disposal of mining waste (DOE-NETL 2006). Overall, estimates of consumptive water use range from 10 to 150 gallons per ton of coal produced (Gleick 1994). Lovelace (2009) reports a range of 50 to 59 gallons per ton for typical U.S. production, or about 22.7 gal/GJ on average.

Coal mining can also have important water quality impacts locally. Surface materials are removed to access the coal, disrupting local surface and groundwater flow regimes. Further, large amounts of water are often extracted from underground formations during the mining process. This "produced" water is typically highly contaminated with volatile compounds, salts, ammonia, and hydrogen sulfide, and if not properly managed can be harmful to water quality. Rainwater runoff from mine sites can contaminate surface flows with heavy metals and alkyls and can lower freshwater pH, leading to acid mine drainage (AMD) (McMahon and Price 2011). As the most significant water quality impact from coal production, AMD can pollute surface flows for up to 40 years and groundwater for over 100 years. In the United States alone, over 9,000 miles of rivers and streams are currently polluted due to AMD from abandoned coal mines (Bian, Inyang et al. 2010).

Natural gas is easier to extract than the other fossil fuels discussed above, as it co-occurs with these other in a pressurized, gaseous state. Gleick (1994) estimates that its extraction requires only small amounts of water (1.6 gal/GJ) for preparation of the drilling fluid. A notable exception to this low water demand is natural gas

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[13] According to figures released by the US Energy Information Administration (2012), extraction of coal east of the Mississippi river, rose continuously until 1990 and then fell continuously thereafter, while total U.S. coal extraction has continued to rise to date.

derived through hydraulic fracturing, or "fracking." In this process, fluids are forced into the ground to create cracks and pores in the rock through which gas can escape. The creation of one gas well in a coal bed through fracking requires between 50,000 and 350,000 gallons of water. Fracking to create a well in a shale formation requires much more water – up to 5 million gallons (US EPA 2010).

6.4.2 Thermoelectric power generation

In the United States, about 89% of total electricity comes from thermoelectric power plants (Torcellini, Long et al. 2003). According to figures released by the United States Geological Survey, thermoelectric power facilities accounted for 41% of all water withdrawals in the U.S., or about 73.4 trillion gallons, in 2005 – more than any other sector. 71% of this withdrawal was from freshwater sources, with the remainder coming from brackish or ocean water. This water was used to generate 3.19 million GWh of electricity, or about 23,000 gallons per MWh.[14] (Kenny, Barber et al. 2009).

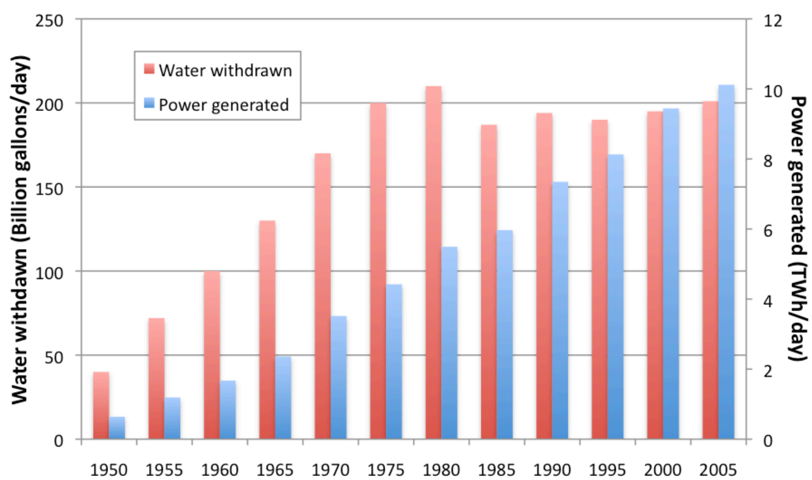

Figure 2-4: U.S. water withdrawals for thermoelectric power generation (data from Kenny, 2009; US EIA, 2012)

Although these huge volumes of water are withdrawn from waterways for thermoelectric power generation, very little of that water is actually consumed. On average in the United States, only about 2.5% of water withdrawn by thermoelectric power plants is consumed (Torcellini, Long et al. 2003). The primary use of water in thermoelectric power facilities is for cooling systems.[15] As a result, the type of cooling employed - more than any other factor – affects both the water withdrawal

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[14] While power use has increased over the past decades, efficiency has improved as well. In 1950, for example, the equivalent figure was 63,000 gal/MWh. This efficiency gain is expected to continue, with average withdrawal for thermoelectric power generation expected to drop to 9,400-12,000 gal/MWh by 2030 (McMahon, Price et al. 2011).

[15] Water is also typically used as the working fluid in thermoelectric power systems, which operate by boiling it for steam to drive turbines. However, this water is part of a closed system, wherein the steam is cooled and returned to the boiler. As a result, water demand for this component of the plant operation is small enough to be negligible as compared to cooling water requirements.

and consumption associated with power generation. Other important factors affecting water use dynamics are plant and cooling system age, thermal efficiency, and water source (Macknick, Newmark et al. 2011).

About 45% of U.S. electric power generating capacity in 2010 used once-through cooling systems (EIA 2011). In these systems, water is withdrawn from a surface flow near the facility, run through a heat exchanger to cool the condenser water, and then returned to its source. However, when the cooling water is discharged, its temperature is typically about 10-20° F higher than it was when withdrawn (Vine 2010). This waste heat causes down-stream evaporative losses, amounting to between 1% and 2.5% of the withdrawn water (Goldstein, Smith et al. 2002; Torcellini, Long et al. 2003). Thermal pollution can also have ecological implications, altering fish migration patterns and aiding the establishment of invasive aquatic species (McMahon and Price 2011).

Closed-loop cooling systems do not expel cooling water after it is circulated through heat exchangers. Instead, the cooling water is itself cooled through evaporation from cooling ponds or towers, and is then reused. These systems, therefore, only need to withdraw enough water to make up for evaporative losses. However, because they rely on evaporation to lower the cooling water's temperature, closed-loop systems consume more water than once-through systems. As a result, recirculating cooling withdraws 10-100 times less water than once-through cooling, but consumes about twice as much (Solley, Perlman et al. 1998; Torcellini, Long et al. 2003; USDOE 2006; Macknick, Newmark et al. 2011). Water scarcity and stipulations in the Clean Water Act have led the U.S. electricity generation infrastructure to move towards closed-loop cooling in order to reduce total withdrawals and heat pollution. As a result, consumptive water use per unit of electricity produced has increased since 1980 (US DoE 2006).

Dry cooling systems are the most water efficient of all, but are much less common. In these systems, the hot condenser water is cooled using a liquid-to-air heat exchanger. This consumes negligible water directly, but can reduce thermodynamic efficiency, resulting in increased water use elsewhere due to increased fuel demand. This inefficiency is particularly acute on hot days, when the temperature of the heat sink rises, thereby creating risks as output can be reduced just as demand peaks (McMahon and Price 2011).

Table 2-4: U.S. thermoelectric power generation capacity by type and cooling system in 2010 (US EIA 2011). "Other" includes petroleum and biomass electricity.

	Coal	Natural Gas	Nuclear	Other	Total
once-through	139208	60627.6	46670.2	21530.6	268036.4
closed-cycle	176509	7082.6	46336.7	10698.0	313089.9
dry air/other	1536	79546.2	300.7	92.7	9012.0

6.4.3 Nuclear Power

Water is used throughout the nuclear power life cycle, from uranium mining through milling, refining, enriching, and power generation. Nuclear fuel is mined in 14 countries worldwide, with three countries – Australia, Canada, and Kazakhstan, producing over half of all uranium produced globally (US IEA/NEA 2010). Mining of nuclear fuel can be a water-intensive process, with water being used for dust control, ore beneficiation, and revegetation of abandoned surface mines (Gleick 1994). In situ recovery involves using the native groundwater along with some solvents to dissolve the Uranium out of underground formations. In total, mining and fuel processing activities use between 45 and 150 gallons of water for each megawatt-hour of electricity generated by nuclear power (Vine 2010)

In the generation phase of nuclear power production, the primary water demand is for cooling. The cooling systems of nuclear plants operate in a manner fundamentally similar to that described above in the general thermoelectric power section, though nuclear plants often withdraw more water per unit of power produced than fossil-fired thermoelectric power plants. The average U.S. nuclear power plant with a once-through cooling system withdraws between 25,000 and 60,000 gallons of water per MWh produced and consumes about 400 gallons. Plants making use of closed-loop cooling technologies withdraw much less water – from 800-1100 gallons per MWh – but consume about 720 gallons (Vine 2010).

6.4.4 Hydroelectric power

Hydroelectricity is the dominant form of renewable energy globally, and its generation has doubled since 1990 (McMahon and Price 2011). As of 2008, hydroelectricity accounted for 16.3% of total electric power generation, with the highest penetration occurring in Norway, which generated 98.5% of its power through hydroelectricity (IEA 2010).

Table 2-5: Hydroelectricity production in the five nations with the most hydroelectric power generation. Data from the International Energy Agency (2010).

Country	% of global hydroelectricity	Total hydroelectric power generation (TWh/yr)
China	17.8	585
Canada	11.5	383
Brazil	11.2	370
United States	8.6	282
Russia	5.1	167

While thermoelectric generation is the dominant source of power in the U.S., hydroelectricity makes up 9% of total production nationwide. This fraction is much higher in the West, where it makes up over 11% of production in California, and almost 90% in both Oregon and Washington (Torcellini, Long et al. 2003). This preponderance of hydroelectric power in the Northwest along with the use of closed-loop cooling at most thermoelectric power facilities in the Southwest due to that region's water scarcity, has led to the western U.S. accounting for only 16% of total U.S. water withdrawals for thermoelectric power (Kenny, Barber et al. 2009).

The water running through the turbines in hydroelectric power generation is not considered consumed, since it flows into waterways where it can provide environmental services or perform any number of domestic, industrial, or agricultural functions downstream. However, there is some consumptive use, as the reservoirs that are constructed in order to generate hydroelectricity lose water through evaporation from their surfaces.[16]

Torcellini et al. (2003) estimate additional evaporation based on surface area and reservoir location for some major hydroelectric facilities in the western US. For example, they estimate that Lake Powell, the reservoir in Utah and Arizona created by Glen Canyon Dam loses about 350 billion gallons of water per year to evaporation. The flow interruption caused by hydroelectric facilities can also have an important negative impact on aquatic ecosystems.

Table 2-6: Consumptive water use for electric power generation by source. All values in gal/MWh

	Thermoelectric Power - CA$^{\alpha}$	Thermoelectric Power - USA$^{\alpha}$	Hydroelectric Power$^{\beta}$	Nuclear Power$^{\gamma}$
Total$^{\delta}$	2,054	570	4,827	650
Ground	2	2	0	2
Surface	6	406	4,827	463
Saline	2,046	162	0	185

$^{\alpha}$ Withdrawal estimates from Kenny et al. (2009). Consumptive fraction from Torcellini et al. (2003)
$^{\beta}$ Estimate from Torcellini et al. (2003); $^{\gamma}$ Estimate from Gleick et al. (1994); $^{\delta}$ Source breakdown derived from Kenny et al. (2009)

6.4.5 Solar and Wind

Approximately 3% of U.S. electric power production in 2011 came from solar and wind – a 16-fold increase since 2001 (US EIA 2012). Typically, these renewable technologies use only the very small water volume necessary for occasional washing of the infrastructure. The exception to this is the array of technologies referred to as concentrating solar power (CSP). These technologies focus solar energy on a small, central system, which then generates electricity using a heat engine (typically a steam turbine). These systems require cooling processes similar to those used in thermoelectric power facilities. Between these cooling systems, and the frequent mirror washing that is necessary to maintain performance, CSP systems can be more water-intensive than conventional thermoelectric power – using up to 800 gal/MWh (McMahon and Price 2011).

[16] These losses cannot necessarily be attributed entirely to hydroelectricity, as the reservoirs serve other purposes as well, including flood control, water storage, and recreation.

The primary water use for solar PV and wind energy production is in the upstream production and fabrication processes.[17] Production of solar photovoltaic cells requires very little water, while wind turbine manufacture can be slightly more water intensive because of the steel required. Moreover, solar cells and wind turbines can be produced in regions where there are sufficient water resources and shipped to drier regions such as deserts where they may be most productive. There is very little water use associated with the operation of these facilities.

6.5 Comparing technologies

The water intensities of the different energy life cycles discussed above vary based on factors such as specific technology employed, location, and timing. However, by comparing average values – as in Table 2-7 – we can shed some light on the relative effect on water resources from each of these energy carriers. Due to its agricultural water demand, bioenergy is orders of magnitude more water intensive on a consumptive basis than any of the other major energy carriers in use today.

Table 2-7: Consumptive water use for energy carriers covered in this chapter - reported as gallons of water used per GJ of *final* energy produced. These are average values and are derived from the more detailed analysis presented in the preceding sections.

	Energy Carrier	Feedstock cultivation	Refining	Total
Liquid Fuels				
	U.S. corn ethanol	19,100[α]	45[β]	19,200
	U.S. cellulosic ethanol	11,400[α]	118[γ]	11,500
	Ethanol from waste biomass	-	118[γ]	118
	Gasoline from U.S. crude	25.1	1.5[δ]	37.5
	Gasoline from Canada oil sand	20.4[ε]	1.9[δ,ε]	36.2
Electricity		**Fuel extraction**	**Power generation**	
	Coal	81[η]	158[ι,χ]	231
	Natural Gas (combined cycle)	4.8[δ]	158[ι,χ]	163
	Hydroelectricity	-	1340[χ]	1340
	Nuclear	27.2[λ]	181[χ,λ]	208

[α] (Fingerman, Torn et al. 2010); [β] (Wu, Mintz et al. 2009); [γ] (Schnoor 2007); [δ] (Gleick 1994); [ε] (Peachey and Eng 2005); [η] (Lovelace 2009); [ι] (Kenny, Barber et al. 2009); [χ] (Torcellini, Long et al. 2003); [λ] (Vine 2010)

[17] As indicated in section 2.1, most of the sections in this chapter do not consider upstream water use in processes such as facility and infrastructure construction. These processes are mentioned here in regard to solar PV and wind power because they make up a non-negligible part of the life cycle water use for these water-efficient technologies. For fossil energy technologies, which typically have high variable water use for operation, embedded water in infrastructure makes up a much smaller portion of life cycle use.

CHAPTER 3:

THE WATER FOOTPRINT OF BIOFUELS: A CALIFORNIA CASE STUDY

3.1 Introduction[18]

Faced with erratic petroleum prices, security concerns, and climate change, governments across the globe have implemented policies aimed at increasing the share of biofuels and other alternative fuels in the energy mix. These policies have helped lead to a six-fold increase in biofuel production over the past decade. Biofuels currently represent about 2% of the global transport fuel and are projected to continue expanding rapidly (US EIA 2012). One such policy is California's Executive Order S-01-07 (Schwarzenegger 2007), the Low Carbon Fuel Standard (LCFS). The LCFS exemplifies a global trend towards use of life cycle assessment (LCA) in policy design to directly target a reduction in fuel GHG intensity.

One flaw in this model, however, is that many of the myriad environmental and social impacts of biofuel expansion are ignored or even exacerbated when policies focus solely on GHG intensity. The initial LCFS technical and policy documents (Arons, Brandt et al. 2007; Brandt, Eggert et al. 2007) are explicit about their exclusive concern for fuel's GHG profile. The technical analysis, part I states:

> *This report addresses only the climate change impacts of fuels, and does not address other public health and environmental impacts…Many of these issues will become more important if biofuel production and use expand, and they are critical to the long-term viability of all energy resources.*

One such impact of potential concern is the probable effect of this and other renewable fuel policies on water resources. This consideration has received little attention despite the fact that biomass energy carriers are usually orders of magnitude more water-intensive than conventional energy sources (King and Webber 2008; Chiu, Walseth et al. 2009; Gerbens-Leenes, Hoekstra et al. 2009; Service 2009; Wu, Mintz et al. 2009; Fingerman, Torn et al. 2010; Scown, Horvath et al. 2011). In implementing this standard, the California Air Resources Board must consider the non-climate implications of its policy options, including the potentially significant effect on water resources.

In this chapter I detail my research into the effect on California water resources of increased ethanol production under the Low Carbon Fuel Standard (LCFS). I provide context through some background on California agriculture and water resources, and develop a methodology for estimating water consumption in California biofuel production. I then project the water resource implications of some scenarios for biofuel production under the LCFS and conclude with policy recommendations for

[18] This chapter is derived in large part from the following published work:
Fingerman, K. R., M. S. Torn, M. H. O'Hare, and D.M. Kammen. (2010). "Accounting for the water impacts of ethanol production." Environmental Research Letters **5**(1): 014020.

the Air Resources Board in implementing the standard with consideration for water sustainability.

3.2 Background – Water Resources

3.2.1. California water resources

California receives about 200 million acre-feet of precipitation and in-flow in the average year, which makes up the state "water budget." However, this flow varies greatly from year to year (California Department of Water Resources 2005)

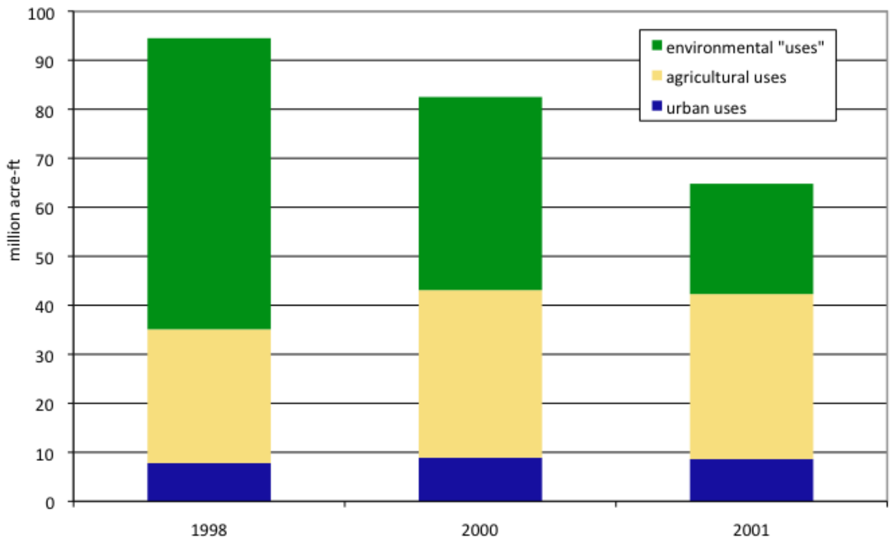

Figure 3-1: Use of California dedicated water supply in varying rainfall conditions. Data from California Department of Water Resources (2005)

Figure 3-1 presents the uses of water in wet, average and dry years. During the three years shown, the state received 171%, 98% and 72% of average rainfall respectively (California Department of Water Resources 2005). While urban water use remained largely stable, agricultural diversions rose both in real and relative terms when water was more scarce, presumably because reduction in rainfall led to increased need for irrigation.

On average, water use in California results in an annual 1.6 million acre-ft[19] budget shortfall. This shortage is made up largely through overdraft of groundwater, a resource that provides 30% of annual water consumption (Howitt, Sunding et al. 2003; Sumner, Bervejillo et al. 2003)

3.2.2. Water and California agriculture

California's agriculture sector – the fifth largest in the world – is entirely dependent upon the availability of water, both through rainfall on fields and through irrigation. As a result, 84% of the developed water in the state is used to irrigate its 9.68

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[19] An acre-ft is a volumetric measure equal to the amount of water required to cover an acre of land at a depth of one foot. (1af = 325,851.43 gal = 1,233,482.1 L.)

million acres of agricultural land (Howitt, Sunding et al. 2003). Figure 3-2 details the total acreage and average applied water per acre for the top 15 crops (by market value) in California for 2010.

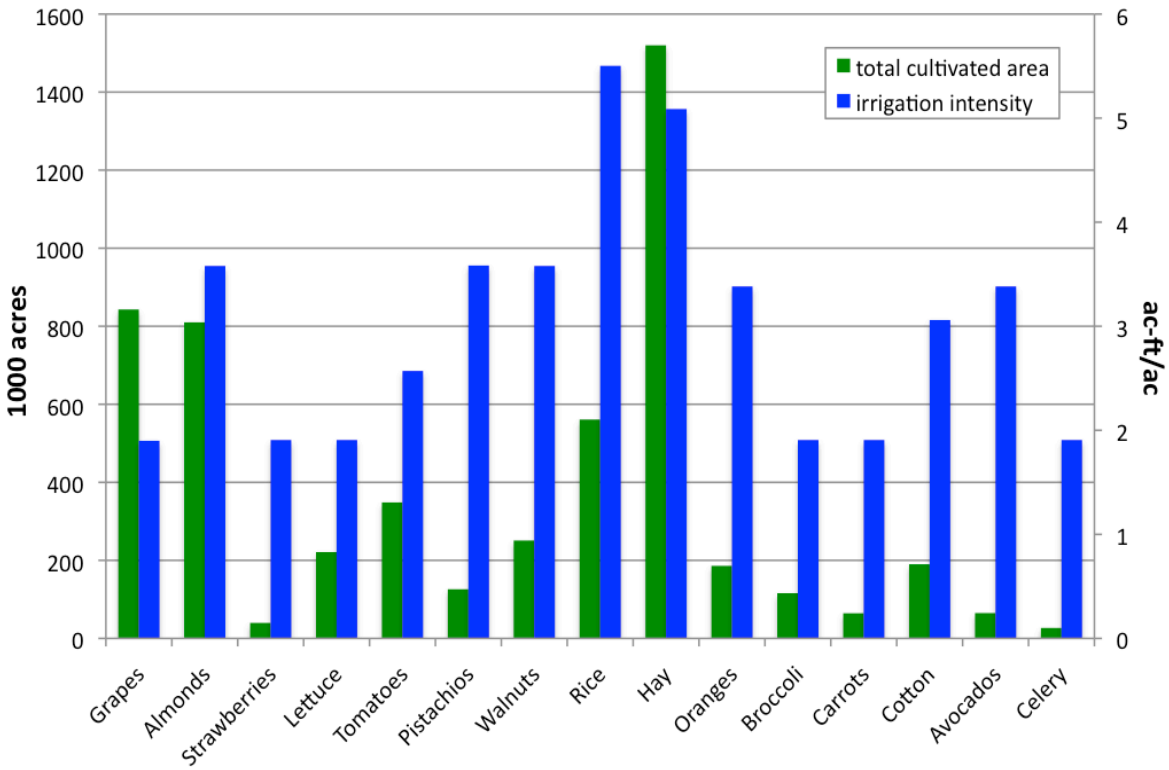

Figure 3-2: Land and water use for the top 15 crops in California (by total cash receipts in 2010 – in descending order from left to right). Data derived from California Department of Water Resources and California Department of Food and Agriculture.

Due to variation in plant physiology and cultivation practices, cultivation of different crops requires vastly different amounts of water. Furthermore, water required to grow the same crop in different climates also varies. For example, water applied to alfalfa (by far the largest user of irrigation water in California) ranges from 2.7 ac-ft per year in Placer County in the Sierra-Nevada Mountains to 6.6 ac-ft per year in the Imperial Valley at the southeast corner of the state.

3.2.3. Water use in biofuel production

Water is consumed[20] all along the biofuel supply chain. Figure 3-3 shows the major uses of water necessary for the agricultural and industrial phases of biofuel production.

[20] As discussed in Chapter 1 of this dissertation, use of the term “consumption” is complicated by the fact that most of the processes being considered here do not actually destroy water molecules. I rely here on a commonly used definition of water consumption: water is considered consumed when it is removed from the usable resource base for the remainder of one hydrologic cycle. Evaporation, therefore, is considered a form of consumption. Although the water has simply changed phases, we do not control where evaporated water will fall next, so the water is functionally lost to the system.

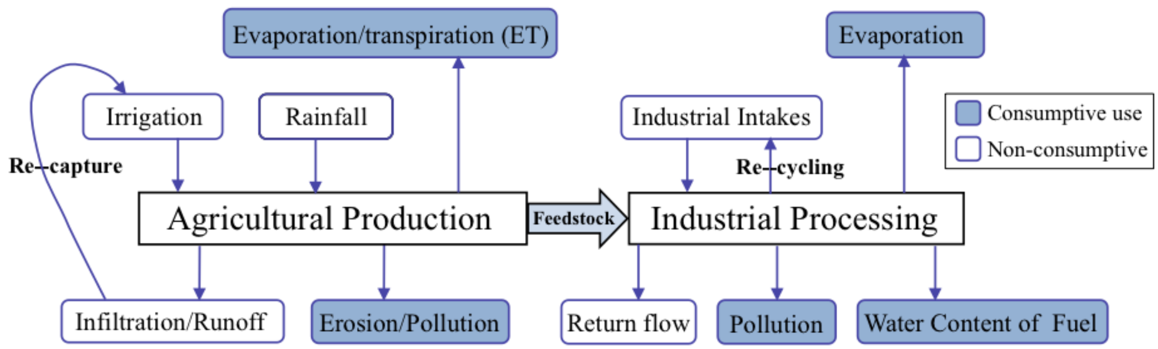

Figure 3-3: Schematic of water uses in the biofuel life cycle. Flows of water both into and out of the bioenergy production and processing system are represented. Source: Fingerman et al. (2010)

Two types of consumption are considered in this analysis[21]:

Crop Evapotranspiration

The largest consumptions of water on the planet are evaporation and transpiration (essentially productive evaporation through plant tissues) – collectively termed evapotranspiration (ET).

Industrial/biorefinery consumptions –

Water is consumed in industrial processes through uses such as cooling and incorporation into finished products.

Many studies of the water resource impacts of biofuel production only consider uses at the biorefinery (Keeney and Muller 2006; Schnoor 2007). While industrial consumption is important, especially for local water resources, this method does not account for agricultural water consumption and therefore does not fully characterize the life cycle effect of biofuels on water resources.

Some recent studies (King and Webber 2008; Chiu, Walseth et al. 2009; Service 2009; Wu, Mintz et al. 2009; Scown, Horvath et al. 2011) include agricultural consumption in their analyses, but only account for water that is applied to fields through irrigation. Considering only irrigation water implies that the 80% of global agriculture, and 77% of US corn production (USDA 2008), that is exclusively rainfed consumes no water. While irrigation water is a vital resource, rainwater is also of value, and if not devoted to biofuel feedstock production could be allocated to other productive uses, to environmental services, or to reservoir and/or groundwater recharge (Molden 2007).

In studies considering all evapotranspiration (Gerbens-Leenes, Hoekstra et al. 2009), lack of spatial resolution leads to an illusion of uniformity in what is actually a very heterogeneous system. Owing to differences in crop physiology, management,

[21] Pollution can also be considered a form of consumption. However, this consumption is of a fundamentally different nature than ET and industrial consumption as the water is degraded rather than consumed. As a result, pollution is outside the scope of this study.

energy yield, and climate, the amount of water required to produce a gallon of biofuel varies spatially. Furthermore, because water availability also varies over space and time, the implications of consuming a given volume of water will not be uniform. This is an important difference between water resource impacts and GHG emissions, which have essentially uniform and widespread effect wherever and whenever they occur.

In this chapter I develop a quantitative framework for evaluating the water resource effects of biofuel expansion for use in LCA and policy analysis. I take as a case study the state of California, which has been a leader in the development of LCA-based fuel policies and which, through its wide variety of agricultural systems and climate types, allows us to draw robust and broadly applicable conclusions. California is also facing severe water supply and allocation challenges. I propose a list of quantitative metrics to enable rigorous analysis of the water use associated with bioenergy production and policy for a variety of concerns and contexts.

3.3 Methods

We used the Penman-Monteith model, a well-established crop evapotranspiration model that uses plant physiology and climate data to calculate water consumption on a daily time-step (Allen, Pereira et al. 1998). These calculations were performed at a county-level resolution in order to capture the spatial heterogeneity of water resource requirement for bioenergy production in California. The calculated ET was then incorporated into a life cycle assessment of biofuel water consumption.

3.3.1. Evapotranspiration modeling

Crop water consumption is estimated in many studies by calculating ET using the Penman-Montieth model developed by the UN Food and Agriculture Organization (Allen, Pereira et al. 1998; Chapagain and Hoekstra 2004; Hoekstra and Chapagain 2007; Gerbens-Leenes, Hoekstra et al. 2009). As shown in Equation 1, Penman-Monteith estimates ET as the product of a reference crop evapotranspiration (ET_o) and a crop coefficient (K_c) (Allen, Pereira et al. 1998).

$$(1) \qquad ET_c = K_c \times ET_o$$

where:

ET_c	=	Total evapotranspiration (mm day^{-1}) from crop *c*
K_c	=	Physiological crop constant varying from 0 to 1
ET_o	=	Reference crop evapotranspiration (mm day^{-1})

K_c accounts for the effect of characteristics such as crop height, surface coverage, and albedo. These characteristics distinguish a crop from the reference surface of uniform grass at 0.12m in height. The evapotranspiration of this reference surface (ET_o) is dependent upon climatic factors such as temperature, solar radiation, wind speed, and relative humidity and is characterized by equation 2.

$$ET_o = \frac{0.408\Delta(R_n - G) + \gamma\left(900/_{T+273}\right)U_2(e_s - e_a)}{\Delta + \gamma(1 + 0.34U_2)} \quad (2)$$

where:

Δ	=	Slope of the vapor pressure curve (kPa °C^{-1})
T	=	Average air temperature (°C)
γ	=	psychrometric constant (kPa °C^{-1})
e_s	=	saturation vapor pressure (kPa)
e_a	=	actual vapor pressure (kPa)
R_n	=	net radiation at the crop surface (MJ m^{-2} day^{-1})
G	=	soil heat flux (MJ m^{-2} day^{-1})
U_2	=	wind speed at 2 m (m s^{-1})

We applied the Penman-Monteith model, to calculate crop-embedded water, using the Consumptive Use Program (CUP) – the Penman-Monteith model parameterized and refined for the California context by the California Department of Water Resources. This model has been validated using nine years of empirical data from the instrumentation network of the California Irrigation Management Information System. (Orang, Snyder et al. 2005). CUP uses measured monthly solar radiation, maximum, minimum, and average temperature, dew point, and wind speed to compute ET_o for climatic regions within California. K_c values are adapted from values published by the United Nations Food and Agriculture Organization (Doorenbos and Pruitt 1984; Allen, Pereira et al. 1998) for analogous crop types. We did not apply a stress coefficient (K_s). In other words, we estimated ET for standard conditions, which assumes crops achieve full production for the given climate (Allen, Pereira et al. 1998) Under water stress, both ET and yield will be reduced, such that the ET/yield ratio reported here is not expected to be highly sensitive to water stress above the wilting point.

3.3.2. Case study feedstocks

Biofuel production in California could be increased through a variety of pathways. Conventional energy crops commonly grown in California include corn, other grains, and sugar beets. Beyond these conventional biofuel feedstocks, there are the “second generation” feedstocks, such as biomass from dedicated energy crops, agricultural residues, and municipal solid waste. Pursuit of each of these feedstocks and production pathways would have distinct implications for California water resources.

Conventional feedstocks

Corn grain is the primary feedstock for biofuel production in the United States. In California, plantings rose almost 25% from 2006 to 2011 (USDA 2008), with the majority of this acreage producing forage and silage to supply the state’s livestock operations. Sugar beets also hold a great deal of potential as a biofuel feedstock in California owing to yields that are among the highest achieved anywhere. While only 25,000 acres were dedicated to sugar beet cultivation in the state in 2011, this figure could easily expand to exceed the 1970 high of 300,000 acres (USDA 2008).

Biomass energy crops

According to the UC-LCFS study, California could have sufficient feedstocks for production of over 1 billion gallons of lignocellulosic biofuel per year by 2020 (Arons, Brandt et al. 2007; Brandt, Eggert et al. 2007). While production at this scale depends upon development and commercialization of technologies that are still largely experimental, lignocellulosic biofuels are likely to play a role in the low-carbon energy future of California.

Purpose-grown biomass crops such as the perennial grasses miscanthus and switchgrass (*Panicum virgatum*) hold promise as future sources of bioenergy while also providing environmental benefits ranging from carbon sequestration (Tilman, Hill et al. 2006) to reduction in erosion and chemical runoff (Helmers, Isenhart et al. 2006; Koo-Oshima 2007). However, no sufficiently broad field tests of miscanthus and switchgrass to date provide reliable data on these crops' water productivity. Furthermore, existing crop evapotranspiration models are calibrated for current crop systems and have not yet been applied to most biomass crops. As a result, this analysis makes use of two hypothetical biomass crop feedstocks – one low-yield and one high-yield – using outside data to project biomass yield, water consumption, and ethanol productivity.

The low-yield biomass (LYB) crop is modeled here on grassy fodder crops (hay and haylage) currently grown in the state. Similar to lignocellulosic feedstock crops, fodders have been bred and cultivated to maximize total plant biomass rather than one specific plant product as is the goal with most crops. The average productivity of these crops is approximately 8.2 dry tonnes of biomass per hectare annually – similar to the yields anticipated from low-input high-diversity grasslands (Tilman, Hill et al. 2006).

The high-yield biomass (HYB) crop in this analysis is modeled as producing 20 dry tonnes of biomass per ha on average annually after Williams (2006) with the relative yields in various California regions modeled on common biomass crops currently grown in the state. This is comparable to yields predicted for energy crops such as miscanthus (Hastings, Clifton-Brown et al. 2009). The water consumption dynamics of these hypothetical HYB crops are modeled using biomass crop water use efficiency values developed by Berndes (2002).

3.3.3. Case study exclusions

Although an estimated 33.6 million tons of biomass residues and waste products could be collected annually in California for conversion to liquid fuel (California Biomass Collaborative 2005), we did not analyze the embedded water in fuels derived from these feedstocks. Quantifying the embedded water in these fuels hinges on the difficult question of co-product allocation, and there is no established method or empirical constraints upon which to base an approach.

Sugarcane, while a major feedstock for ethanol production globally, is not grown widely in California and so is not analyzed here. Biodiesel is also not considered here as it plays a much smaller role in state transport fuel projections than does ethanol. Furthermore both soy and canola, the two major agricultural feedstocks for biodiesel, are grown in such small quantities that their area is not reported in the USDA agricultural census.

Finally, we did not account for indirect effects of feedstock production on water resources in which water consumption is altered far from the production site by market-mediated land-use change (Searchinger, Heimlich et al. 2008; Melillo, Reilly et al. 2009; Hertel, Golub et al. 2010). A standard method for their quantification has not been developed and is the subject of much debate.

3.3.4. Analytical approach

We carried out these analyses at a county scale. The study included all counties in California, though we only considered individual feedstocks in those counties where they (or comparable crops in the case of biomass feedstocks) are currently grown. We assume that field crops – specifically corn, wheat, rice, sorghum, barley, oats, cotton, beans, and fodder crops – will be displaced for feedstock production. These are low-value crops, and are therefore more likely to be replaced by biofuel feedstocks than are higher value crops such as fruits and vegetables. Furthermore, these field crops are annuals, so the land is available the following year at no loss as opposed to being tied up in a long-term investment such as an orchard.

County-level agricultural production and per-acre productivity data used in this study are drawn from the USDA National Agricultural Statistics Service's (NASS) agricultural census. NASS county-level production data are also used for weighting of statewide averages. Agricultural inputs are modeled after estimates published by the University of California Cooperative Extension service. Data on refining processes and outputs are drawn from the EBAMM model(Farrell, Plevin et al. 2006), the GREET model (Wang 2009), and the NREL model biorefinery (Aden, Ruth et al. 2002).

3.4 Results

3.4.1. Water consumption

In all of the feedstock cases studied, the agricultural production phase represented more than 99% of life cycle water consumption on average with biorefineries consuming less than 1% of total ethanol embedded water. Our analysis shows a clear difference in fuel embedded water among the feedstocks modeled. Table 3-1 shows the average embedded water in ethanol from each of the feedstock crops weighted by county feedstock production (Equation 3).

(3) $$\sum_{i,c} ET_{ic}\left(\frac{y_{ic}}{Y_c}\right)$$

where:

i	=	county
c	=	feedstock crop
ET_{ic}	=	evapotranspiration of crop c in county i (L H_2O/L ethanol)
y_{ic}	=	yield of crop c in county i (T)
Y_c	=	total yield of crop c in California (T)

The results of the analysis described above on are presented on statewide weighted-average basis in Table 3-1.

Table 3-1: Water embedded in ethanol produced in California (L H_20 per L EtOH). The values in this table are production-weighted averages of county-scale water intensity estimates.

Feedstock	**ET** (L H_20/L EtOH)	**Refinery** (L H_20/L EtOH)
Corn Grain	1,533	3.6[a]
Sugar Beets	1,271	3.6[a]
Low-Yield Biomass	1,301	6[b]
High-Yield Biomass	916	6[b]

[a]Wu, Mintz et al (2009)
[b]Aden, Ruth et al. (2002)

The production-weighted averages presented in Table 3-1 are derived from a dataset exhibiting substantial county-level heterogeneity in both yield and crop ET. The statewide average embedded water in ethanol from the different feedstocks studied varied by more than 50% due to plant physiological properties and feedstock cultivation region. Figure 3-4 illustrates the breadth of values seen for these crop characteristics.

The feedstocks studied varied in both their water consumption and their yield per hectare. This result leads to both spatial and feedstock-related variation in water consumption per MJ of fuel produced. The variation stems from climatic factors such as temperature, wind, and relative humidity within the study area as well as from physiological differences among the feedstock crops studied. Water consumption ranged from less than one acre-foot per year for cultivation of low-yield biomass in Modoc County to more that 4 acre-ft per year for cultivation of sugar beets in Imperial County. Ethanol yield ranged even more dramatically, from less than 1000 L per ha for low-yield biomass in Sierra County to almost 14,000 L per ha from high-yield biomass in Imperial County.

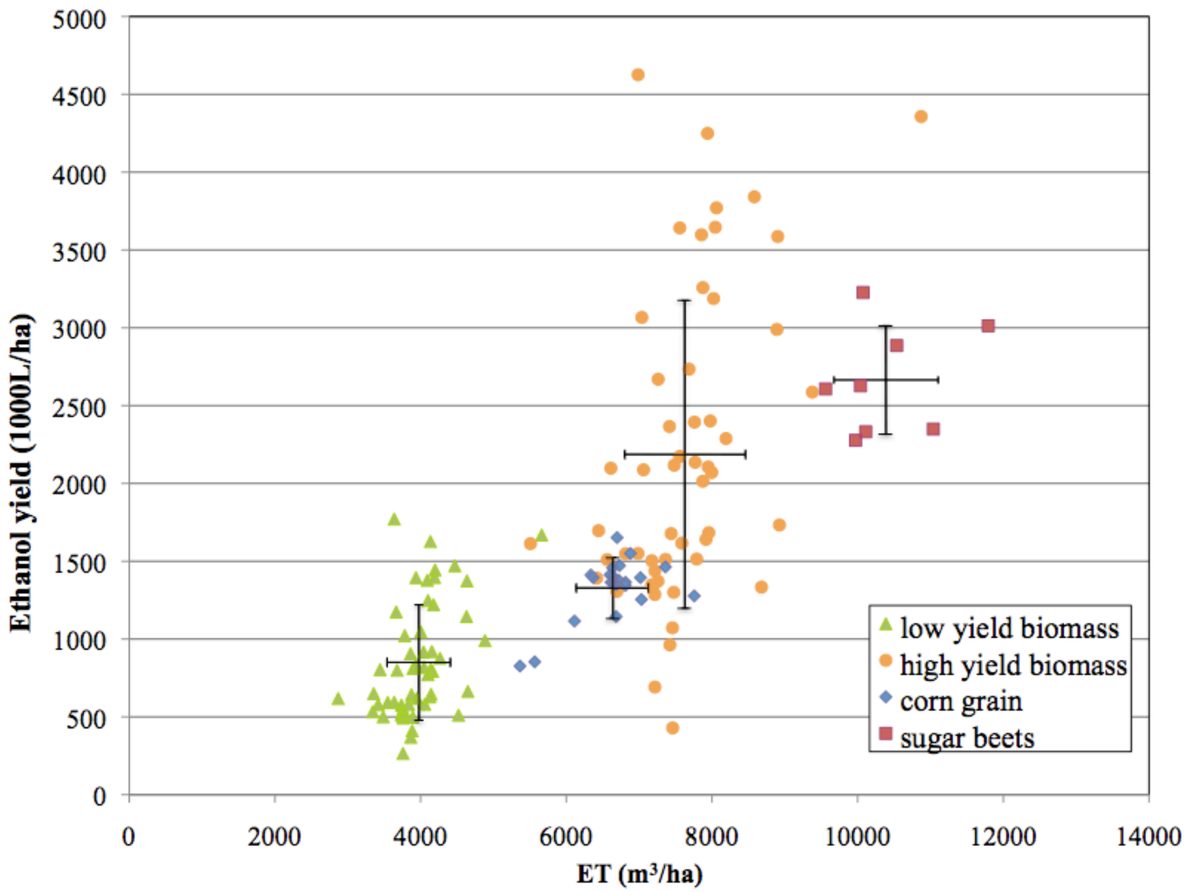

Figure 3-4: ET and yield by county. Each data point represents a feedstock in a county where it is currently grown. The error bars show the standard deviation about the mean among counties for each feedstock type; the bars cross at the mean value for both parameters.

This variation in yield and ET among counties creates patterns in the geographic variation of water consumption. Figures 3-5(a) and 3-5(b) focus on the low-yield biomass feedstock because it is grown in every county in the state. Figure 3-5(a) presents the water consumed per liter of fuel from each of these places, while Figure 3-5(b) shows the amount consumed per hectare cultivated. The contrasting patterns of these two maps stem from the fact that while more water is consumed in cultivation of this crop in the southern reaches of the state, those areas are also more productive per unit consumption. Imperial County in Southern California was found to have both the highest water consumption of any county in the state and the greatest yield. This is of particular interest in light of the fact that more than 70% of average runoff occurs north of Sacramento while the southern part of the state accounts for over 75% of total water demand (Sumner, Bervejillo et al. 2003).

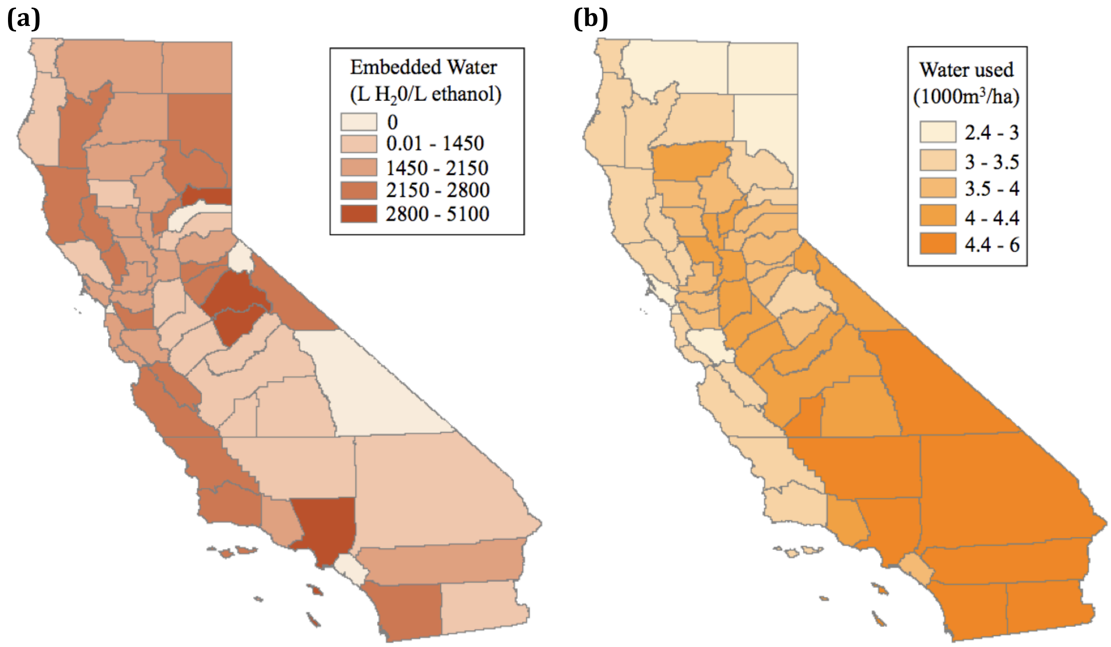

Figure 3-5: (a) Embedded water in ethanol from low-yield biomass feedstock. (b) Yearly per hectare water consumption for cultivation of low-yield biomass feedstock.

Implementing the LCFS could have a significant aggregate effect on California's water resource base. I assess this effect here based on an estimated 830.4 million gasoline gallons equivalent (GGE) of in-state ethanol production. This figure assumes that 40% of the biofuel consumption projected by Brandt et al. (2007) is produced in California in accordance with the target set in Executive Order S-06-06 (Schwarzenegger 2006). Table 3-2 lays out the projected aggregate consumption of in-state water resources from meeting the LCFS solely through the use of each feedstock studied here.

Table 3-2: Net water consumption (as evapotranspiration) relative to total supply

Feedstock	Total area (million acres)	Consumption (ac-ft/ac)	Total consumption (million ac-ft)	% of average irrigation[a]	% of average total supply[a]
Corn	2.67	2.19	5.86	17.23%	2.93%
Sugar Beet	1.36	3.58	4.86	14.28%	2.43%
LYB	3.61	1.38	4.97	14.63%	2.49%
HYB	1.32	2.65	3.50	10.29%	1.75%

[a] California water plan update, 2009

While the total water demand for ethanol can be substantial, the net effect will depend on the prior status of the land being employed in feedstock cultivation. If regional average field crops were replaced with biofuel feedstocks, there would be a *decrease* in total water demand statewide, as some of those displacements would occur on land previously occupied by very water-intensive crops such as rice and alfalfa. If these heavily irrigated crops were *preferentially* displaced, the water savings would be significant. If, however, the bioenergy targets were met through expansion of irrigated agriculture, the net increase in water demand would be considerable.

3.5 Discussion

The amount of water required to produce ethanol from purpose-grown feedstocks in California was found to range from under 500 liters of water per liter of fuel to over 3500. In comparison, production of an equivalent (by energy content) volume of gasoline requires from 2.1 liters of water for conventional petroleum crude to almost 14 liters for fuel from tar sands (Gleick 1994; King and Webber 2008).

It would be overly simplistic, however, to compare these figures directly – concluding that petroleum fuels are over 100 times more water-efficient than ethanol. While industrial processing of biofuels and petroleum fuels requires comparable amounts of water, biofuels also require water for the cultivation of feedstocks, whereas the feedstock for petroleum refining is extracted from the earth. Although over 99% of the water "embedded" in an agricultural biofuel is used in feedstock cultivation, this consumption is much more spatially diffuse than the refining of the feedstock into biofuel. Feedstocks, grown over a large area, are concentrated in one locale to be refined into biofuel. As a result, in some cases effects on the *local* resource base may be larger from the industrial phase of the biofuel life cycle than from the agricultural phase. Furthermore, a new refinery, whether for biomass or petroleum refining, represents a *new* demand on water resources. In contrast, feedstock production may result in an increase or a decrease in water demand depending on what, if anything, is replaced.

Location of water consumption is also an important consideration. For example, cultivation of rainfed crops in tropical regions where water is plentiful may not result in water scarcity although it may consume a great deal of water. In contrast, use of scarce groundwater for refining petroleum in arid regions where it is produced might put excessive strain on that resource base even though it may be comparatively "efficient" in simple terms of water consumption per unit energy. This is an important difference between analyses of life cycle water consumption, and those of GHG emissions; use of water has very different implications in different contexts, whereas GHG emissions have essentially the same effect wherever they are produced.

3.5.1. Functional units for analysis

Life Cycle Assessment (LCA) requires that impacts be normalized and reported in terms of a common "functional unit." Current literature on bioenergy water consumption implicitly assumes that impacts on water resources can be normalized and compared through the use of a single metric. Metrics considered have included gal H_2O applied per distance traveled (King and Webber 2008; Mishra and Yeh 2011; Scown, Horvath et al. 2011), L H_2O applied per L ethanol (Chiu, Walseth et al. 2009), m^3 H_2O consumed per GJ ethanol (Gerbens-Leenes, Hoekstra et al. 2009), and gal H_2O applied per gal ethanol (Wu, Mintz et al. 2009; Fingerman, Torn et al. 2010).

One of the important messages of our research is that in moving the energy system to use water more responsibly, different measures of consumption will be important in different contexts and that there is no one consistently superior functional unit for analysis.

While the “water footprint” of a biofuel can be calculated in terms of L H_2O consumed per MJ, in assessing the effects of a production system we may at times be concerned with ascertaining the volume of water consumed per *hectare* used to grow feedstock. These two metrics can lead to very different pictures of the production system as is illustrated by Figures 3-5(a) and 3-5(b). If our interest lies in optimizing the former, Figure 3-5(a) would indicate that we should focus production in the southern reaches of California. If, however, we are interested in the equity of resource distribution, or in minimizing the energy-intensive pumping of water across great distances, Figure 3-5(b) would indicate that production in the northern reaches of the state is the more viable option.

The dichotomy illustrated in Figures 3-5(a) and 3-5(b) reveals the importance for water resource analysis of choosing the proper functional unit for analyzing the issue at hand. Table 3-3 presents many of the metrics that might be of value and concern in evaluating the water resource implications of biofuel production. It lays out the analytical utility of each metric, as well as the context and scale in which each might be used as a basis for optimization and the assumptions embedded in doing so.

A critical issue relating to these metrics is whether the sustainability of water resource management hinges on the metric of L H_2O *consumed* per unit product (in this case MJ of ethanol) or on L H_2O *applied* per unit product. The latter metric’s focus only on irrigation application implies that rainfed crops consume no water, which is clearly not the case. Rainwater is a valuable resource that could be dedicated to other productive use, to environmental services, or to aquifer recharge if not consumed in biofuel feedstock cultivation (Molden 2007).

On the other extreme, some researchers argue that evapotranspiration should be the primary or exclusive metric of concern, since irrigation water can run off or infiltrate to rejoin the exploitable resource base. While this may be true on a macro scale, in many localities water that is applied inefficiently is lost to productive use, causing irrigation demand to be a more important concern than total ET.

No unit is *de facto* best for analysis and optimization of water resources in the bioenergy system. Instead, local nuance must be accounted for if policies are to adequately address this important implication of bioenergy expansion.

Table 3-3: Functional units for Life Cycle Assessment of water resource effects of biofuel feedstock production. Included are the *analytical utility* of each metric, as well as the *context* and *scale* in which each might be used as a basis for optimization and the *assumptions* embedded in doing so.

Unit	Analytical utility	Context	Scale	Embedded assumptions
		For use as a basis for optimization		
L H_2O consumed/ MJ fuel produced	Compare to overall water resource base with other energy carriers	Water Productivity ($ per m^3 consumed); "water footprint"	Basin, region	All water consumed is of equal importance
L H_2O applied/ MJ fuel produced	Compare to alternate uses of the resource	Irrigated regions with stressed irrigation resources; Energy inputs to fuel production	Field, basin, irrigation district	Irrigation water is of primary significance; rainfed crops consume no (important) water
L H_2O consumed/ha	Compare with precipitation to determine irrigation demand or environmental flows	Where resource is limited for environmental flows or groundwater recharge	Field, basin	Water conservation is more important than maximizing derived value
L H_2O applied/ha	Compare to regional average water demand	Where many farms/farmers rely on the same resource	Basin, irrigation district	Equity of distribution is of greater value than overall optimization of water productivity
Pollution impact/ MJ fuel produced	Evaluate life cycle pollution impact insofar as this can be quantified	Resource/environment stressed by industrial effluent or agricultural runoff	Field, basin, region	Pollution impact can be quantified per unit output in a meaningful way

3.5.2. Recommendations

Biofuel policies aimed at reducing life cycle GHG emissions are a step in the right direction, but many policy makers are beginning to recognize that a fuel with a life cycle GHG footprint better than that of its petroleum analog is not necessarily environmentally benign (Schlegel and Kaphengst 2007; Verdonk, Dieperink et al. 2007; Hecht, Shaw et al. 2009).

In some cases, optimizing a biofuel production system with regard to greenhouse gas emissions could increase the strain on water resources. For example, the developing understanding of market-mediated or “indirect” land use change (Searchinger, Heimlich et al. 2008; Hertel, Golub et al. 2010) may cause production of biofuel feedstocks to trend away from currently cultivated land. This could mean extensification of agriculture, potentially bringing new strain on irrigation water resources. Furthermore, the imperative to increase yields in order to minimize cropland required for biofuel could cause growers to irrigate cellulosic crops such as miscanthus, which are currently grown in rainfed conditions.

Water resource implications of bioenergy policies should be considered in the rule-making process to ensure that these policies do not drive changes that will put undue stress on water supply. I recommend the following actions and further research to analyze and elaborate them into practical form:

- **Implement a water accounting system.** Our analysis shows the feasibility of calculating the water embedded in biofuels from different feedstocks grown in various regions.
- **Establish water intensity regulations for low-Carbon fuels.** Calculated or reported water consumption and application for biofuel production should be incorporated into regulatory frameworks with incentives for implementing best management practices.
- **Regulate siting and design of biorefineries.** While water consumption by biorefineries is a relatively small portion of total biofuel embedded water, it may have a large local effect. Careful siting and design of biorefineries will minimize conflicts between different water uses as well as ensuring that the waste streams from plants cause the least possible harm to the environment and human health.

CHAPTER 4:

IMPACT ASSESSMENT AT THE WATER-BIOENERGY NEXUS

4.1 Introduction[22]

During recent years, concerns about petroleum price volatility, energy security, and climate change have led many governments to institute policies promoting bioenergy for a variety of uses. The resulting expansion in biomass use for energy, and the attendant shift in agricultural and forestry activity, has raised a number of environmental and social concerns, ranging from potentially increased greenhouse gas (GHG) emissions to labor rights abuses, deforestation, and reduced food security. Another important concern – one that has received little attention though it could be the "Achilles heel" of biofuel production (Keeney and Muller 2006) – is the fact that biofuel is very water intensive relative to other energy carriers. Increased water demand from a growing bioenergy industry may place considerable additional stress on available water supplies.

Differing patterns of population growth, lifestyle changes, pollution of available water, and climate change will present different scenarios in every region, but water users, managers, regulators, and planners are challenged to meet growing water needs virtually everywhere. Population growth, economic development, and changing diet are projected to drive a 70-90% increase in demand for food and feed by 2050, resulting in a comparable expansion in global water demand barring major changes in production patterns and water productivity (Molden 2007). By 2025, over half of the world's population is expected to be living under conditions of "low or catastrophically low water supply (Shiklomanov 2000)." This scarcity is intimately connected to both poverty and health concerns for human populations, as well as hindering the proper functioning of local ecosystems. Bioenergy expansion must be seen in this wider context of water scarcity, particularly in its competition for water needed to grow food.

Energy and water are deeply interrelated, though different energy carriers have very different "water footprints" depending on geographic location and the processes involved in their production. In the United States, thermoelectric power generation accounts for 49% of total freshwater withdrawals (Kenny, Barber et al. 2009). Most of this water, however, is only used briefly for once-through cooling. On a consumptive basis,[23] biofuels derived from purpose-grown agricultural feedstocks are the most water intensive of all major energy types, usually by at least an order of magnitude (King and Webber 2008; Chiu, Walseth et al. 2009; Dominguez-Faus, Powers et al. 2009; Gerbens-Leenes, Hoekstra et al. 2009; Wu, Mintz et al. 2009; Fingerman, Torn et al. 2010). Their influence on water resources and the wider hydrologic cycle depends on where, when, and how the biofuel feedstock is produced.

[22] This chapter is derived in large part from the following published work:
Fingerman, K. R., G. Berndes, S. Orr, B. D. Richter, and P. Vugteveen (2011). "Impact assessment at the bioenergy water nexus." Biofuels, Bioproducts and Biorefining **5**(4): 375-386.

[23] Use of the term "consumption" is complicated by the fact that most of the processes being considered in this chapter do not actually destroy water molecules. We rely here on a commonly used definition of water consumption: water is considered consumed when it is removed from the usable resource base for the remainder of one hydrologic cycle. Evaporation, therefore, is considered a form of consumption. Although the water has simply changed phases, we do not control where evaporated water will fall next, so the water is functionally lost to the system.

Among different bioenergy supply chains, across the spectrum of feedstocks, cultivation systems, and conversion technologies, there is a great deal of heterogeneity in total water demand. Where fuel made from irrigated crops can require large volumes of water, use of agricultural or forestry residues as bioenergy feedstocks does not generally require much additional land or water. Rainfed feedstock production does not require extraction from water bodies, but can still affect downstream water availability by redirecting precipitation from runoff and groundwater recharge to crop evapotranspiration. The net hydrologic effect depends on what types of vegetation (if any) are removed to make room for bioenergy feedstock production and how land use and management are changed.

Further variation derives from *spatial* heterogeneity in the water impact of bioenergy production. A given production activity will consume varying amounts of water depending upon where and when it occurs, and the consumption of a certain amount of water has very different social and ecological consequences depending upon the state of the resource base from which that water is drawn.

Policy, infrastructure, and business decisions related to bioenergy expansion can carry with them either ecological and social benefit or detriment, so we must consider the probable consequence of proposed activities holistically. To the extent that bioenergy projects cause unwelcome effects on an array of social, political and environmental concerns, trade-offs between benefits and costs will need to be managed.

This chapter seeks to inform these decisions in light of the complexity of the bioenergy-water nexus. The goals of this chapter are to:

1. *Describe the ways in which tools commonly applied to assess water impacts of bioenergy can be insufficient for the purposes of business or government decision-making.*
2. *Lay out an assessment framework that should be followed in order to make informed decisions at the bioenergy-water nexus.*
3. *Describe considerations and tools that can be usefully brought to bear on these questions.*
4. *Indicate some location-specific impacts that may be of concern in introducing or expanding bioenergy production, but that may not be revealed even through rigorous and nuanced application of common analytical approaches.*

4.2 Common analytical tools at the bioenergy-water nexus

Water Footprint[24] (WF) accounting, and more generally water life cycle assessment (LCA) as they have been applied to bioenergy (De Fraiture, Giordano et al. 2008; King and Webber 2008; Dominguez-Faus, Powers et al. 2009; Gerbens-Leenes, Hoekstra et al. 2009; Wu, Mintz et al. 2009; Fingerman, Torn et al. 2010), are currently insufficient in their treatment of ecological and social impacts for the purposes of decision-making and mitigation of detrimental effects. LCA was designed, and has been most commonly used, to evaluate industrial products and activities, which generally require relatively low volumes of water.

[24] Use of the term "water footprint" is confounded by the fact that different researchers apply the term in different ways. Some use the term to signify any life cycle water impact. For the purposes of this chapter, however, we will refer to these analyses simply as "water LCA" and will confine our use of the term "water footprint" to the analytical approach pioneered by Arjen Hoekstra and colleagues (Hoekstra et al, 2007) which is most comparable to the Life Cycle Inventory phase of water LCA.

LCA tools have therefore not been developed to sufficiently address water impacts of activities (Milà i Canals, Chenoweth et al. 2009; Berger and Finkbeiner 2010).

Furthermore, many LCA tools were developed largely for use in calculating life cycle greenhouse gas (GHG) emissions, which have a functionally uniform impact wherever they occur. Water consumption, however, has implications that vary greatly depending on what resource base is affected, the previous state of that resource, the opportunity cost of its use, and the location and timing of the use in question.

Water use has been considered in the life cycle inventory (LCI) phase of LCA analyses of various goods (Hoekstra and Chapagain 2007), as well as in a few studies of bioenergy (Berndes 2002; De Fraiture, Giordano et al. 2008; King and Webber 2008; Gerbens-Leenes, Hoekstra et al. 2009; Wu, Mintz et al. 2009; Fingerman, Torn et al. 2010). This work generates salient figures – establishing, for example, that producing a liter of corn ethanol requires 10-324 L of water (Service 2009). Such quantified water demand can be useful in high-level policy making, particularly when used to compare different activities in a given region. However, water volumes alone are of little use in efforts to minimize impacts, since a given level of consumption in a watershed with still-abundant water supplies can be expected to have far less impact than the same consumption in a watershed experiencing severe water scarcity. The above LCA studies have not made localized assessments, distinguishing among types of water use, the sources from which it was drawn, or accounting for local conditions.

Location-specific information on water resource use and impacts is essential to inform responsible decision-making in relation to specific bioenergy projects, or more comprehensive agriculture development plans. This information is not provided by WF studies as conventionally applied,[25] or by water LCA studies, which tend to focus analytical rigor only on the inventory phase of the analysis. These tools measure the amount of water used in production of various goods, but lack proper characterization of relative water scarcity and the opportunity cost of water use to conduct meaningful life cycle impact assessment (LCIA). As Berger and Finkbeiner (2010) state, where intensities are calculated based purely on consumption inventory, they "can be meaningless or even misleading with regard to impact assessment."

4.3 Framework for assessing water sustainability of bioenergy

Because of the significant interface between water and energy, responsible policy or industry development for bioenergy requires careful consideration of water requirements and potential effects on the local and regional water resource base as well as ecological health and other uses of water. This chapter seeks to inform this process by addressing the question of *what we need to assess* in order to determine whether a proposed bioenergy development would have a positive, negative, or neutral effect on the state of water resources in a given area.

In making or advising decisions in this space, the following questions should guide our deliberations (Figure 4-1):

[25] Current WF best practice, as laid out the "Water Footprint Manual: State of the Art 2009" (Hoekstra et al, 2009) includes a process of "water footprint assessment." However, such a complete analysis has yet to emerge in the WF literature for biofuels.

1. What do we need to know about the water intensity of an operation or activity that is proposed in a given region/watershed? How can we measure this?
2. What do we need to know about the state of water resources as well as societal and environmental water needs in the region or watershed? How can we measure this?
3. What important potential outcomes for water cannot be captured in these measurements? How can we characterize these?

This chapter seeks to answer these questions, offering an *assessment framework* that operators and policy makers can use as one component in their assessment of the sustainability of proposed activities.

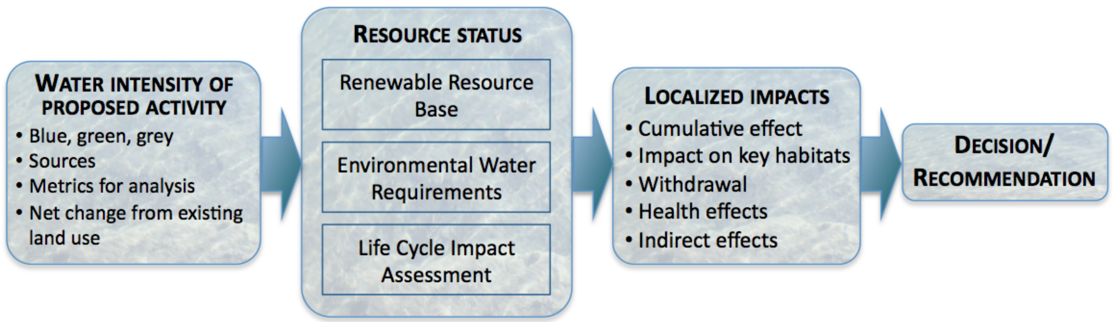

Figure 4-1: Assessment framework: key elements for evaluating the water resource impacts of bioenergy expansion.

4.4 Measuring water intensity

This section lays out the types of water use that should be considered for a water footprint assessment or in the analogous inventory phase of water LCA for biofuels. Water is consumed at multiple points along the biofuel supply chain. Figure 3-3 shows the major uses of water necessary for the agricultural and industrial phases of biofuel production. A comprehensive life cycle inventory (LCI) would account for many or all of these flows, as well as any others specific to the supply chain in question.

Water resource use and impacts can defy easy quantification for a number of reasons:

1. A given activity can require vastly different amounts of water in different locations and at different times due to climatic differences and other factors.
2. Different water sources are not easily comparable; withdrawals of freshwater from surface flows will have a different effect than groundwater pumping, rainfall or the use of brackish water.
3. Even where the same resource *type* (e.g. river flow) is used or polluted, the impact of that activity can vary widely depending on the context of that use, where and when it occurs, and the current status of the affected resource base.

Surface water flows in rivers and lakes, as well as groundwater aquifers provide important ecosystem services, as well as being available for a variety of human uses. This resource is collectively termed ***blue water***, and in the case of bioenergy production, its use can include consumption of water in cooling a refinery as well as any water applied as agricultural irrigation. Some studies on the water impact of biofuels have focused only on blue water consumption at the biofuel production facility. (Keeney and Muller 2006) However, since the bulk of the water use in most bioenergy supply chains occurs due to feedstock cultivation activities many researchers turn their attention toward irrigation in their analyses of water use for bioenergy (King and Webber 2008; Chiu, Walseth et al. 2009; Dominguez-Faus, Powers et al. 2009; RSB 2010).

Not all of the water applied as irrigation is actually consumed; some of it runs off or infiltrates and is later available for other productive agricultural or ecological uses. Similarly, much of the water that is consumed by plants is not applied through irrigation, but is instead ***green water*** – naturally available through rainfall in the plants' root zone. If not devoted to biofuel feedstock production, green water can go to other productive uses – to cultivation of another crop, to environmental services, or to groundwater recharge (Hess 2010). Furthermore, considering only blue water, applied as irrigation, implies that the 80% of global agriculture that is rainfed(Serageldin 2001; Molden 2007) consumes no water.

The most common approach for estimating green water consumption is through modeling total evapotranspiration (ET). This is usually done using the Penman-Monteith model (Allen, Pereira et al. 1998), which incorporates plant physiology as well as climatic factors such as solar radiation, wind speed, humidity, and temperature. Calculated ET can be used along with estimates of effective (usable) rainfall and applied irrigation to calculate blue and green water consumption. Variations on this approach have been used for bioenergy by Berndes, (2008) De Fraiture *et al*.,(2008) Fingerman *et al*.,(2010) Gerbens-Leenes *et al*.,(2009) and Wu *et al*.(2009)

Pollution can be considered a consumptive use of water (sometimes called ***grey water***), since it removes a certain volume from being later utilized productively. A "grey water footprint" is considered to be that volume of freshwater required for dilution of total pollutant load to below a defined ambient water quality standard. Some studies have made an effort to quantify the fertilizer intensity of bioenergy supply chains (Mubako and Lant 2008; Dominguez-Faus, Powers et al. 2009; Fingerman, Kammen et al. 2009) as well as to assess the macro-scale environmental effects of fertilizer use for bioenergy expansion. (Donner and Kucharik 2008).

The impact of a given amount of water consumption or pollution will depend primarily upon the resource base being affected. For example, the implications of pumping renewable groundwater are different than those of extracting fossil groundwater, diverting water from a river, or from an irrigation canal, though each would be termed "blue water" use. For this reason, any accounting method should, to the extent possible, disaggregate different sources. Similarly, impact will vary greatly depending on when and where water is consumed and returned to source. It is essential, therefore, that analysis be performed at the greatest feasible spatial and temporal resolution.

Even working to include the various types of water use and sources described above, and at the necessary spatial resolution, results can still be misleading. LCAs are generally built around a single "functional unit," in which the impacts in question are measured. Choice of this functional unit for analysis can greatly alter the perceived patterns of impact. In their case study of California ethanol production detailed in chapter 4 of this document, Fingerman, Torn, et al. (2010) found that very different conclusions could emerge depending on whether water impact is quantified per liter of fuel produced or per hectare devoted to feedstock production.

4.4.1 Getting to "impact"

A variety of impacts can result from consumption or degradation of freshwater resources. These can include diminished ecosystem functioning from reduced natural flows as well as impacts on human health and well-being due to poor water quality, quantity, or lack of access (Milà i Canals, Chenoweth et al. 2009). These impacts do not depend, however, solely upon the absolute quantity of water consumed or degraded by an activity. More important is the *net consumption*, taking into account the water intensity of any activities or land uses affected or displaced by the project in question.

Even having conducted a comprehensive, disaggregated, and spatially detailed water footprint assessment or life cycle inventory, accounting for considerations described above, we will not have adequately characterized the *impact* of the activity in question. Life cycle impact assessment (LCIA) relates the LCI data to "the potential human health and environmental impacts of the environmental resources and releases identified during the LCI (ISO, 1998)." For water, it is impossible to understand these impacts without assessing the current state of the resource base in which the expected change would occur.

4.5 Characterizing the local water resource base

Characterization of impact reflects the significant complexities to be found in natural resource systems and their use by society. Values placed on different potential uses and functions, perceptions of significance, and changing water use priorities are individually and culturally subjective. At the core of understanding impacts in water resource systems is recognition of this complexity and an effort to characterize impacts within local physical and social contexts. Some ecological and social conditions can become impaired at very low levels of water resource disturbance, while others may remain undisturbed until water conditions have been altered significantly.

Individual water LCI or WF results are essentially incommensurable, as the impact of water consumption varies greatly depending on what resource base is affected, the previous state of that resource, as well as the location and timing of the use in question. In life cycle impact assessment (LCIA), characterization (or weighting) factors are derived, whereby these different consumption values can be summed and compared across resource bases and locations. In the case of greenhouse gas LCA, global warming potential (GWP) is often used as a characterization factor to normalize across different types of GHG emissions. The most useful characterization factors for water LCA are derived from metrics of strain on the water resource base. A review of the literature reveals a set of approaches for assessing the status of a local water resource base.

4.5.1 Characterizing scarcity

Perhaps the most basic metric for evaluating water scarcity is that of water resources per capita (equation 1). WRPC values have been used, for example, to set annual threshold values for *water stress*—less than 1,667 (or 1,700) m^3 per capita, *water scarcity*—less than 1,000 m^3 per capita, and *absolute water scarcity*—less than 500 m^3 per capita (Falkenmark 1986). Though it is a standard indicator of water scarcity, common applications of WRPC have not considered intra-annual variation of water availability, differences in water use patterns between countries, or any in-stream uses (Raskin, Gleick et al. 1997).

$$(1) \qquad WRPC = \frac{R}{C}$$

where:

WRPC	=	*Water resource per capita*
R	=	*Total renewable water resource in area of interest [m^3 $year^{-1}$]*
C	=	*Population drawing on the resource*

The water use per resource indicator (WUPR - equation 2) could be more useful, since it considers all human water uses, and reports them as a proportion of the total renewable resource base (Raskin, Gleick et al. 1997). Whereas WRPC captures only the amount of water available to each person independent of any activities, WUPR begins to account for how human extraction has strained resources.

$$(2) \qquad WUPR = \frac{U}{R}$$

where:

WUPR	=	*Water use per resource*
U	=	*Total human uses in area of interest [m^3 $year^{-1}$]*
R	=	*Total renewable water resource in area of interest [m^3 $year^{-1}$]*

One metric drawing on the WUPR approach is the Water Stress Index (WSI)[26] presented by Pfister *et al* (2009). The WSI (Pfister) was introduced as a characterization (or weighting) factor for life cycle impact assessment, and has been used to incorporate impact assessment into traditional WF analyses (Pfister, Koehler et al. 2009; Ridoutt and Pfister 2010). Pfister *et al*. (2009) innovate in inserting some local nuance into their calculated WUPR (which they refer to as withdrawal to availability - WTA). They apply a variation factor to account for climate variability and for storage and flow regulation in managed watersheds. Furthermore, the authors recognize that water stress does not occur linearly with increased WUPR; low levels of withdrawal might not generate much stress, and in very highly stressed watersheds further withdrawal may not have much marginal effect. To account for this, Pfister *et al*.(2009) apply a logistic function to translate WUPR values to their Water Stress Index, which is reported on a scale from .01 to 1.

$$(3) \qquad WSI(Pfister) = \frac{1}{1 + e^{-6.4*WUPR*VF}(99)}$$

where:

WSI(Pfister)	=	*Water stress index*
VF	=	*Variation factor - from precipitation variability*

The WSI(Pfister) and other common WUPR metrics use withdrawal as their gauge of water use. Withdrawal, however, is not an adequate proxy for consumption. For instance, use of water for thermoelectric generation consumes very little (~2%) of the water withdrawn, whereas irrigation use typically consumes 40-50% (Solley, Pierce et al. 1998). This means that WUPR-based tools, as commonly applied, can produce highly misleading results.

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[26] Some confusion has emerged from the fact that both this metric, and another similar one called the Water Stress *Indicator* (introduced by Smakhtin *et al.*[49]) are both abbreviated as "WSI." To distinguish, these will henceforth be referred to here as WSI (Pfister) and WSI (Smakhtin).

Furthermore, to be useful as an indicator of freshwater ecosystem health, an LCA tool must account for environmental flow requirements. None of the above metrics incorporates this at present.

4.5.2 Environmental flows

Based on principles of ecosystem science and integrated water resource management (IWRM) (e.g. Gleick 2000), the concept of "environmental flow" has emerged – defined by the Brisbane Declaration (2007) as the "quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems." This concept recognizes that there is a limit to human alteration of natural hydrologic conditions, beyond which a water resource suffers unacceptable (and possibly irreversible) damage to its ecosystem functions and the ecosystem services[27] the river provides to humans (Bunn and Arthington 2002; Poff and Zimmerman 2010). Maintenance of environmental flows should be viewed as both a goal and a primary measure of sustainability in river basins (Dyson, Bergkamp et al. 2003; Hirji, Davis et al. 2009; Richter 2010).

While bioenergy currently accounts for a small percentage of total water use, its expansion has been implicated in water-related environmental impacts ranging from streamflow reduction to changes in soil-level vapor processes (De Fraiture and Berndes 2009). Withdrawals from surface- and groundwater resources may lead to reduced streamflows in rivers, groundwater decline, and wetland drainage. Other impacts include eutrophication and toxicity effects that can result from the application of fertilizers and agrichemicals (Smeets, Bouwman et al. 2009). Conversely, biomass can be cultivated in such a way as to offer water resource benefits. Some plants can be cultivated as vegetation filters for treatment of nutrient-bearing water such as pretreated wastewater from households and runoff from farmlands (Dimitriou and Rosenqvist 2010). Groundcovers and vegetation strips can be located to limit wind and water erosion, reduce evaporation of surface runoff, trap sediment, enhance infiltration, and reduce the risks of shallow landslides. (Berndes, Börjesson et al. 2008; Dimitriou, Busch et al. 2009; Rowe, Street et al. 2009)

On larger scales, changes in land use and cover – such as reforestation of sparsely vegetated lands to provide carbon sinks or produce biomass for energy – may also affect watershed-scale hydrology, causing shifts in evapotranspiration, runoff dynamics, and other hydrologic attributes, potentially altering local climate (Brauman, Daily et al. 2007; Uhlenbrook 2007).

An array of analytical methods and approaches has been developed for assessing environmental water requirements (EWR) for specific rivers or throughout a region. The methods vary in their complexity, their data requirements, and the financial and human resources needed for analysis. The type of approach most appropriate for use depends on the resources available and the objectives of the environmental flow assessment.

At the most fundamental level, and in the absence of detailed ecological data, the "sustainability boundary approach" presented by Richter (2010) provides a framework for defining the desirable variability to be maintained for rivers, lakes, and groundwater. Under this approach, the human-induced changes to hydrologic flows are managed within specified boundaries (Figure 4-2). Richter *et al.* (in review) subsequently suggested that a

[27] Intact aquatic ecosystems can provide to humans a variety of critical services including water purification, flood and drought mitigation, groundwater recharge, and habitat creation, among others.

sustainability boundary of up to 20% augmentation or depletion from the natural condition would generally maintain good-to-excellent ecological health. These boundaries should be dynamic, reflecting changing societal needs, local ecology, or priorities for resource protection and development.

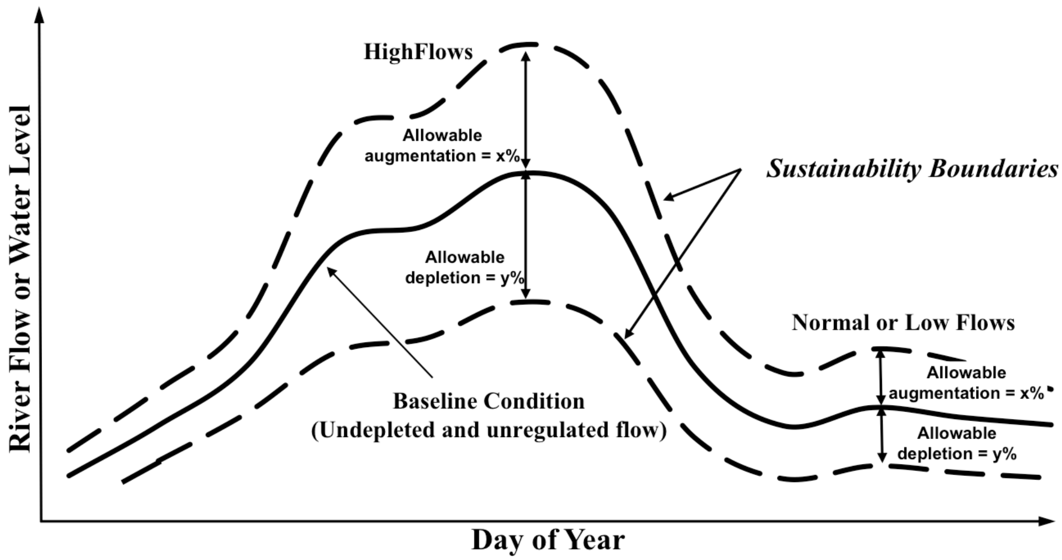

Figure 4-2: Illustration of the Sustainability Boundary Approach (SBA) to maintaining environmental flows. The approach suggests that human-induced flow alterations beyond a certain percentage will likely cause ecological impacts. By managing within sustainability boundaries, ecological impacts can be avoided or minimized. Adapted from Richter (2010).

This type of analysis can be an important first order assessment, especially where data or resources are not available for more in-depth study. At the opposite end of the spectrum lie data and time-intensive approaches incorporating river-specific hydrology, geomorphology, and field ecological study to define environmental water requirements for specific rivers (Tharme 2003; Richter, Warner et al. 2006; King and Brown 2010). Such detailed analysis will be warranted for politically or socially contentious water allocation decision-making.

The recently-developed ELOHA (Ecological Limits Of Hydrologic Alteration) framework (Poff, Richter et al. 2010) offers a compromise between the need for local nuance and the need for an efficient and broadly applicable system. ELOHA analysis involves a number of steps briefly outlined here:

1. The first step in the ELOHA process calls for the building of a hydrologic database to characterize both natural or "baseline" hydrologic conditions as well as "developed conditions" (i.e., current conditions). These baseline and developed hydrographs can be generated using both measured field data and hydrologic modeling.
2. The hydrologic database is then used to calculate the degree of hydrologic alteration that has transpired for each river, or for individual segments of rivers, defined as the percent difference between baseline and developed conditions for a number of ecologically-relevant hydrologic metrics.
3. The baseline hydrologic conditions are used to classify each river or river segment into "river types" that are similar in terms of hydrologic, geomorphic and other environmental features.

4. For each river type, “flow-ecology curves” are developed to describe how ecological conditions are expected to change with increasing degrees and types of hydrologic alteration, using available biological data and the “developed condition” hydrologic data from the hydrologic database.
5. The flow-ecology curves inform decision-making about the environmental flow standards to be set for each river type.

Managing for environmental flows involves setting agreed-upon precautionary boundaries, or benchmarks, beyond which hydrologic regimes should not be altered. The degree of acceptable risk is likely to reflect the balance between the perceived value of the ecological goals and the scientific uncertainties in projected ecological responses to flow alteration (Poff and Zimmerman 2010). Analysis of environmental water requirements (EWR) should inform policy and business decisions at the bioenergy-water nexus; it can be combined with the WUPR-type approaches described above to characterize a basin, assess risk, and project impact.

4.5.3 Incorporating Environmental Water Requirements

The Water Stress Indicator (WSI) of Smakhtin et al. (2004) made a first attempt at estimating environmental water requirements for all of the world’s river basins and using this to characterize risk. They subtract EWR and all other water uses from the available water resources to derive a water stress indicator at the river basin level (equation 4). This is a WUPR-style indicator, which better captures the resources available for human use by considering the available resource base to be only the water left over after ‘reserving’ the necessary flows for ecosystems.

$$(4) \qquad WSI(Smakhtin) = \frac{U}{R - EWR}$$

where:

WSI (Smakhtin)	=	*Water stress indicator*
U	=	*Withdrawal for human use in area of interest* $[m^3\ year^{-1}]$
R	=	*Renewable water resource in area of interest* $[m^3\ year^{-1}]$
EWR	=	*Environmental water requirements*

The WSI (Smakhtin) suffers from the same shortcoming as the rest of the WUPR-based indices in employing only withdrawal as its metric of usage, though it could be adapted to apply to consumptive use as well. Hoekstra *et al.*(Hoekstra, Chapagain et al. 2009) resolve this problem by approaching scarcity from the net consumption framing of the Water Footprint (Equation 5). Their “water scarcity” metric can be applied at any temporal scale, and is designed to be used separately for blue and green water.

$$(5) \qquad WS[x,t] = \frac{WF[x,t]}{R - EWR}$$

where:

WS	=	*Water scarcity (blue or green)*
WF	=	*Total catchment-level water footprint (blue or green)*
x	=	*Watershed of interest*
t	=	*Timescale of interest*
EWR	=	*Environmental water requirements*

The *levels* of environmental water requirements suggested by Smakhtin have been broadly criticized by the ecological science community as being so low as to "…almost certainly cause profound ecological degradation, based on current scientific knowledge" (Arthington, Bunn et al. 2006). These critiques could be addressed by simply increasing the EWR value as a precaution considering uncertainties involved, or by applying a more comprehensive EWR assessment, such as the ELOHA framework detailed above. Furthermore, a more robust analysis using the WSI (Smakhtin) or the blue and green water scarcity metric as a characterization factor for LCIA could employ the variation factor and the logistic curve fitting techniques introduced by Pfister et al. (2009) to account for climate variability and non-linearity of stress effects.

4.5.4 Assessing impact

Tools such as the WSI (Smakhtin) or Hoekstra's blue and green "water scarcity" metric could be used to sort basins, watersheds, or regions into stress classes to evaluate risk or need for further study. In this vein, Hoekstra et al. (2009) propose characterization of "water footprint hotspots" based on spatially and temporally explicit water footprint of an activity and the state of the corresponding resource base. A green water footprint hotspot occurs "when in the catchment re-allocation of the green evaporative flow from natural to productive vegetation takes place at the cost of biodiversity beyond a certain acceptable level." A blue water footprint hotspot occurs when extraction level fails to provide for environmental water requirements. Lastly, a grey water footprint hotspot occurs when pollution violates agreed-upon water quality standards in a catchment.

These tools can also be used directly as characterization factors to give more weight to consumption where strain is greater. This approach has been suggested by Hoekstra et al.(Hoekstra, Chapagain et al. 2009) in the form of "water footprint impact indices," generated separately for blue, green, and grey water impact. More sophisticated characterization factors could account for climatic variability, watershed management, and the non-linear relationships between consumption and impact. Weighted consumption values can then be used to examine specific impacts of greatest concern such as Freshwater Ecosystem Impact (FEI) and Freshwater Depletion (FD) (Milà i Canals, Chenoweth et al. 2009), or human health, resource depletion, and ecosystem change (Pfister, Koehler et al. 2009).

4.6 Evaluating other impacts

The quantitative analytical approaches discussed in this chapter were designed to be broadly applicable, and so may at times fail to capture some location-specific impacts of concern. Other types of impacts may not be easily captured using quantitative metrics. In evaluating a policy or an investment related to bioenergy expansion, decision-makers should consider some of the following location-specific concerns and should work to avoid or to alleviate them where relevant:

- While individual projects may have water impacts well below established thresholds, the cumulative effect might become problematic in regions undergoing rapid change or expansion in energy infrastructure.
- Localized effects may be deceptive. Water consumption for biomass conversion represents less than 1% of the total water footprint for many biofuels (Fingerman, Torn et al. 2010), but because it takes place in a concentrated area, refining might have a

larger local effect than more spatially diffuse – and possibly quite distant – feedstock production.

- A variety of non-consumptive uses, such as once-through cooling, exist in the bioenergy supply chain, but may not be captured by LCA tools. These withdrawals can be important locally, causing ecosystem disruption, heat pollution, or other impacts.
- Impact on key habitats such as aquifer-recharge zones, wetlands, and floodplains can have a large effect throughout a watershed. This local impact might be missed in watershed-scale evaluation (Alliance for Water Stewardship 2009).
- Acute but localized ecological toxicity, eutrophication, or human health effects may result from even small pollution flows.
- Indirect land-use change has been recently recognized as a critical concern for the life cycle GHG impact of bioenergy (Searchinger, Heimlich et al. 2008; Hertel, Golub et al. 2010). Similarly, perturbation of global commodity markets due to bioenergy production could lead to detrimental impact on water resources far from the site of the activity in question. This effect has yet to be studied in any depth.
- Water shortage for human uses does not necessarily derive from absolute scarcity, but can instead be due to social realities such as equity of access, barriers to entry, poor infrastructure, institutional failure, and other considerations that may be affected by bioenergy expansion.
- Water scarcity does not have the same effect in all places. Affected populations vary in their ability to adapt to scarcity through altered lifestyle, development of new resources, and imports of "virtual water."

Assessment and mitigation of these and other localized impacts will require on-the-ground investigation to complement use of the quantitative metrics described in this paper.

4.7 Conclusions

Bioenergy expansion can have significant implications for the state of water resources in the region in which it occurs. As a result, business, policy, and resource management decisions related to bioenergy should take this critical consideration into account. Water resource impacts can defy easy quantification because water consumption varies spatially and temporally, different water sources are not necessarily commensurable, and impact depends on the state of the resource base that is drawn upon. In order to describe impacts, life cycle inventory (LCI) must be comprehensive, accounting for both blue and green water use as well as for pollution effects, varying sources, and the spatial heterogeneity of usage. Furthermore, where possible it should quantify the *net effect* of activities, accounting for the consumption associated with any displaced land uses rather than only quantifying consumption in absolute terms.

Nuanced and disaggregated LCI is an essential component of impact assessment, but does not in itself serve to sufficiently characterize impact. Unlike greenhouse gases, which have a functionally uniform impact wherever or whenever they are emitted, water consumption has implications that vary depending on context. There is no universally suitable quantitative tool by which to characterize the impact of water consumption. However, the most credible approach is to use tools that quantify consumption in the context of any existing stress on the resource base in question while also accounting for the fact that sufficient environmental flows need to remain intact to maintain a stable ecosystem.

This nuanced and comprehensive analysis can require detailed data that may not always be widely available. However, most of the existing research in this space has been conducted at a large scale and with an eye toward general application. Where considering or advising on specific activities, the scale will generally be smaller and the human and financial resources may be available to gather detailed information *in situ*.

Life cycle impact assessment (LCIA) and/or weighted water footprint impact indices can be important tools for identifying regions of concern for green, blue, and grey water impacts. Some local nuance can be lost through this aggregation of detailed information, even when done in the thorough and spatially discrete manner described in this chapter. For this reason, localized concerns including cumulative effects, impact on key habitats, indirect effects, social realities, and resilience to scarcity should be investigated carefully as a complement to this type of quantitative analysis.

Increasing bioenergy demand is a challenge from the perspective of water resources but it is important to address this challenge based on a holistic approach that considers the pressure placed on water by all competing uses. There is considerable scope for improving water productivity in many regions of the world, reducing the amount of water needed for crop production, and leaving more water for other uses, including environmental flows. The integration of bioenergy with food and forestry production presents interesting opportunities in this regard. The tools presented in this chapter can help to identify opportunities to improve water use efficiency and resource management based on development of biomass supply systems as a new element in the landscape.

Impact assessment using the various tools described in this chapter can be an important first step toward optimizing opportunities from bioenergy production while minimizing any detrimental effects on water resources. In many cases, along with working to mitigate their own impacts, large water users should engage with others in watershed-level restoration and governance activities. Finally, most of the concerns raised in this report are not unique to bioenergy, but are instead instances of larger, systemic issues in agriculture, industry, land use, and natural resource management. As a rapidly growing sector, however, bioenergy can serve as a high-profile leverage point, to raise awareness of water-related issues and to implement best management practices where they may not otherwise occur.

CHAPTER 5:

THE WTO AND ITS IMPLICATIONS FOR REGULATING BIOFUEL SUSTAINABILITY

5.1 Introduction

The mandate of the World Trade Organization (WTO), according to its establishing document, the Marrakesh Agreement, is to "[expand] the production of and trade in goods and services, while allowing for the optimal use of the world's resources in accordance with the objective of sustainable development, seeking both to protect and preserve the environment and to enhance the means for doing so (GATT 1994)." This clear tie between trade and environment is reflected in rhetoric coming out of the WTO leadership. In 2007, WTO Director-General Pascal Lamy, speaking to a conference of national environment ministers in Nairobi, stated that "…no one can argue that sustainable development is a choice anymore. It has become a must. Sustainable development should be the cornerstone of our approach to globalization and to the global governance architecture that we create. If I have come to this forum, it is to deliver a message: the WTO stands ready to do its part (Lamy 2007)."

The reality of the WTO's environmental record is, however, slightly less encouraging on the issue of environmental considerations. De Vera (2007) argues, "WTO case law suggests that while sustainability remains a factor to consider, prohibiting factors that have the slightest tinge of protectionism continues to be the primary concern." Even the WTO's Committee on Trade and Environment (CTE) recognizes in a statement of its guiding principles that the "WTO is not an environmental protection agency and that its competency for coordinating policy in this area is limited to aspects of environmental policies related to trade (Hartwick and Peet 2003)." It is clear from this language, that while our major trade-enabling entities are concerned with issues of sustainability, they see them as secondary to their main goal of trade liberalization.

Even our major multilateral environmental agreements recognize the importance of the free trade doctrine to successful international negotiation in the post Bretton Woods[28] era. The Rio declaration, broadly considered the defining international statement on "sustainable development," borrows language from the General Agreement on Tariffs and Trade (GATT), stating that:

> *"States should cooperate to promote a supportive and open international economic system that would lead to economic growth and sustainable development in all countries, to better address the problems of environmental*

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[28] With World War II drawing to a close, delegates from the allied nations met in Bretton Woods New Hampshire In July, 1944 for the United Nations Monetary and Financial Conference. This conference led directly to the establishment of the International Monetary Fund and the General Agreement on Tarriffs and Trade (precursor to the WTO) – institutions designed to facilitate trade liberalization and the implementation of the neoliberal economic philosophy of the era.

degradation. Trade policy measures for environmental purposes should not constitute a means of ***arbitrary or unjustifiable discrimination or a disguised restriction on international trade.*** *Unilateral actions to deal with environmental challenges outside the jurisdiction of the importing country should be avoided" (UNCED 1992 - Principle 12 - emphasis added).*

This phrase, prohibiting "arbitrary or unjustifiable discrimination or disguised restriction on international trade" is drawn *directly* from GATT, and also appears in Article 3.5 of the United Nations Framework Convention on Climate Change (UNCED 1992).

Central to the WTO's efforts in environmental governance is the liberalization of trade in environmental goods and services (EGS). In the case of climate change, this initiative might involve efforts such as reduction or elimination of import tariffs on environmental goods such as wind and hydroelectric turbines, solar water heaters, biogas tanks, and landfill liners for methane collection. A 2007 World Bank study estimated that eliminating tariffs and non-tariff barriers could result in a 14% increase in the trade of clean energy technologies.

Importantly, this EGS structure only touches on those instances in which trade liberalization creates a net benefit to the environment. Biofuels may not be environmentally beneficial, especially if governments are expected to liberalize their biofuel policies by removing stipulations on life cycle greenhouse gas (GHG) intensity or other social and environmental considerations. Advocates of such conditionalities hold that these linkages are beneficial to all players, making trade fairer to importing countries, while improving human development and environmental performance in exporting nations (Charnovitz, Earley et al. 2008).

Most discussions of biofuel governance now recognize that a life cycle GHG accounting system must be implemented and used as the basis for evaluation in order for any biofuel policy to have the desired climate effect. Still further, if we hope to ensure that environmental and social wellbeing are not harmed by our increased biofuel consumption, sustainability standards would need to be in place, conditioning market access to some degree upon environmental and social performance. The question then arises as to whether such policies would hold up under WTO scrutiny.

5.1.1 WTO and Biofuels

A variety of measures are being implemented or considered by governments seeking to influence trade in biofuels. These policies generally focus on one or more of the following three goals:

1. Increase the total amount of biofuel in the energy mix.
2. Decrease GHG-intensity of the fuel mix.
3. Improve the environmental and/or social performance of biofuels being consumed.

Most of the policies that have been implemented are primarily intended to accomplish goal 1, owing to perceived economic, energy security, and environmental benefits. As it has become clear over the past decade that the amount of GHG emissions associated with biofuels varies greatly depending on production methods, policies have expanded to explicitly address goal 2 above. The third goal, while increasingly important, is only addressed nominally in most of the biofuel policies currently on the books.

To accomplish goal 1 above, a broad array of policy tools is being considered or implemented (Table 5-1). These policies, aimed at broadly promoting biofuels, are not likely to elicit disputes in the WTO so long as they do not violate subsidy rules and are not structured to provide preferential treatment to domestic fuels or to those from specific trading partners.

Table 5-1: General policy models for increasing biofuel production and/or market share. Adapted from Howse (2006).

- Mandatory blend level for biofuel in the liquid fuel mix
- Tax incentives including excise tax exemptions or rebates for retailers/consumers, corporate tax incentives for producers
- Specialized financing initiatives for production/distribution infrastructure, including loans, loan guarantees, grants, etc.
- Regulatory incentives such as exemptions or waivers
- Support of R&D, including basic research or demonstration
- Incentives (tax breaks, subsidies, etc) for purchase of biofuel-compatible vehicles
- Government procurement preferences and purchase mandates for fuels or vehicles

In some cases, these policies include various provisions aimed at achieving goals 2 and 3 by making support conditional upon life cycle GHG emissions or other social or environmental performance metrics. The most forceful such model would be an outright ban on import or use of products that do not meet certain GHG or sustainability criteria. Less forceful, and more common, are policies that condition participation in support programs such as tax incentives, subsidies, or target blend levels on such criteria. Such conditionalities are very difficult to enact rigorously and legally, as they depend on an increasingly complex "web of policy considerations and interwoven legal regimes that link the World Trade Organization (WTO), farmers and cattle barons in Brazil, the European Commission in Brussels, and orangutans in Borneo to transportation fuel..." (Payne 2008).

Many biofuel policies will have a trade-distorting effect and so may become a source of dispute within the WTO. Although no dispute has yet been filed, Brazil, Indonesia, Malaysia, and the United States have all indicated that they may bring a dispute in light of various sustainability criteria associated with the EU's Renewable Energy Directive (RED) (Oosterveer and Mol 2010).

A 2007 World Bank study states that "arguably the greatest technical barrier (to trade) in the coming years could be certification of biofuels for environmental sustainability" (Kojima, Mitchell et al. 2007). It is clear that biofuel policies linked to environmental and social performance, while important, will prove difficult to design, in part because their status with respect to WTO disciplines is as yet uncertain. Biofuels were not an important traded good when the WTO was founded, and they have never featured prominently in a dispute, making the precedent difficult to interpret. Without clarification as to the standing of such policies with respect to WTO obligations, this standing may have to be determined through the dispute settlement process.

With this in mind, this chapter sets out to investigate two related issues:

1. Are life cycle GHG-based biofuel policies WTO compliant?
2. Can policies managing the broader environmental and social performance of biofuel production be designed to be WTO-compliant?

5.1.2 Potentially relevant agreements under the WTO

Whereas GATT is a set of rules agreed upon by nations in 1947, the WTO is an institutional body officially established in 1995 to manage implementation and disputes arising from GATT and other related trade agreements. The Marrakesh Agreement, founding the WTO, consists of the original GATT text, as well as annex agreements adding detail on specific issues such as intellectual property, subsidies, agricultural policy, and technical regulations (GATT 1994).

To avoid a violation of its obligations under the WTO, any trade-related action undertaken by a nation[29] must comply with *all* of the agreements relevant to that policy type.[30] Most existing biofuel policies would fall under the purview of two or more of the following five WTO agreements:

1. *General Agreement on Tariffs and Trade (GATT)* – As the overarching agreement, all trade policies must comply with GATT.
2. *Agreement on Technical Barriers to Trade (TBT)* – Technical regulations, such as GHG performance metrics, might be found to fall under the purview of the TBT.

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[29] According to the WTO Dispute Settlement Understanding article 22.9, the Agreements also apply to actions "taken by regional or local governments or authorities within the territory of a Member. When the [Dispute Settlement Panel] has ruled that a provision of a covered agreement has not been observed, the responsible Member shall take such reasonable measures as may be available to it to ensure its observance." This stipulation is important in the case of biofuel policies implemented at the sub-national level. With this in mind, this chapter considers California's Low Carbon Fuel Standard alongside relevant national and supranational policies.

[30] In the interest of judicial economy, dispute panels do not always determine compliance with every relevant agreement. If, for example, a policy falls under the SPS agreement and is found to be in violation of that agreement, the panel may refuse to examine any claims under GATT even though violations of that agreement may also have been alleged in the complaint.

3. *Agreement Sanitary and Phytosanitary Measures (SPS)* – SPS covers measures regulating trade in agricultural products (including perhaps biofuel feedstocks) to avoid risks such as import of pests or disease.
4. *Agreement on Agriculture (AoA)* – Some agriculture-specific policies, including agricultural subsidies, are managed by the AoA.
5. *Agreement on Subsidies and Countervailing Measures (SCM)* – Biofuel subsidy programs, unless explicitly covered elsewhere such as under the AoA, must comply with the SCM.

This chapter investigates existing jurisprudence under these agreements in order to derive probable implications of WTO obligations for biofuel policies.

5.2 Landscape of globally significant biofuel support policies

Policies are being promulgated globally to expand the market share of bioenergy. If comprehensive, global, environmental and social protection policies were in place, biofuel supports could stand alone, leaving their impacts to be efficiently managed and allocated through other instruments.[31] Given that this is not the case at present, protections must be put into place in order to ensure that biofuel supports do not worsen environmental or social conditions globally.

This section discusses some of the most significant biofuel support policies currently in implementation. Specific attention is paid to these policies' varying treatment of GHG emissions and other environmental and social impacts associated with biofuel production. It should be noted that the various social and environmental impacts of concern here are not unique to bioenergy, but are instead instances of larger, systemic issues in agriculture, industry, land use, and natural resource management. However, the rate of expansion in the bioenergy industry is largely the result of government intervention, making it incumbent upon regulators to assess and mitigate any negative implications of their actions.

5.2.1 US EPA – Renewable Fuel Standard (RFS)

Created under the US Energy Policy Act of 2005, and expanded under the Energy Independence and Security Act (EISA) of 2007, the Renewable Fuel Standard (RFS) aims to expand the production and consumption of renewable transportation fuels in the US energy system.

[31] If, for example, a comprehensive global price on GHG emissions were in place, the cost of life cycle emissions would be internalized in the price of fuels. This would lead to efficient emission allocations, wherein very low-GHG fuels could have lower cost of production and only very inexpensive fuel systems could "afford" to have high emissions. Under these circumstances, biofuel support policies would not need to address life cycle GHG emissions directly as the market would address this issue.

The 2010 revision of the RFS (RFS2) redefines "renewable fuel" and lays out, for the first time, four separate categories of renewable fuel based on feedstock and GHG reduction level,[32] setting volumetric requirements for each category:

- *Renewable Fuel* is defined as all transportation fuels (not only motor vehicle fuels) "produced from renewable biomass and that is used to replace or reduce the quantity of fossil fuel present in a transportation fuel" (USEPA, 2010). "Renewable fuel" must have life cycle GHG emissions of at least 20% less than the gasoline or diesel fuel it replaces.[33] All biofuels counted under the RFS must meet this definition of renewable fuel.
- *Advanced Biofuel* is defined as "a renewable fuel other than ethanol derived from corn starch and for which lifecycle GHG emissions are at least 50% less than the gasoline or diesel fuel it displaces." This fuel thus qualifies also as "renewable fuel."
- *Cellulosic Biofuel* is defined as "renewable fuel derived from any cellulose, hemicellulose, or lignin…It must also achieve a lifecycle GHG emission reduction of at least 60%, compared to the gasoline or diesel fuel it displaces." This fuel thus qualifies also as both "advanced biofuel" and "renewable fuel."
- *Biomass-based diesel* includes "any diesel fuel made from biomass feedstocks" as long as it is made from renewable biomass, has a lifecycle GHG intensity at least 50% less than the diesel fuel it displaces, and is not derived from coprocessing of biomass with petroleum feedstock. Like cellulosic biofuel, this fuel will also qualify as both "advanced biofuel" and "renewable fuel."

An important change from the Energy Policy act of 2005 and its RFS to the EISA and its RFS2 is that the latter changed the definition of renewable fuel from ''any motor vehicle fuel that is used to replace or reduce the quantity of fossil fuel present in a fuel mixture'' to its current version, requiring that the fuel be made from "renewable biomass." This distinction allows EISA and RFS2 to introduce constraints in its definition of "renewable biomass"

Rather than create explicit sustainability provisions, as such, the RFS2 avoids some of the environmental and social concerns associated with biofuels by limiting the types of biomass that qualify, as well as types of land from which that biomass can be harvested. EPA has decided not to include rangeland as "agricultural lands" under this definition, because that could open up sensitive ecosystems such as "savannahs, wetlands, deserts and tundra." They also state that "conversion of relatively undisturbed rangeland to the production of annual crops could in some cases lead to large releases of GHGs stored in the soil, as well as a loss of

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[32] Compared with 2005 baseline gasoline

[33] Fuels from production capacity that commenced construction prior to December 19, 2007 are exempt, (or "grandfathered") from the 20% lifecycle requirement for the Renewable Fuel category.

biodiversity, both of which would be contrary to EISA's stated goals (US EPA, 2010)." Similarly, all forest products (such as slash or pre-commercial thinnings) must come from non-federal lands that are not old growth or listed as critically imperiled or rare.

Table 5-2: Revised fuel targets set in the RFS2.

Year	Advanced Biofuels (billions of gallons)			Total renewable fuel (billions of gallons)
	Cellulosic biofuel	Biomass-based diesel	Total advanced	
2010	0.1	0.65	0.95	12.95
2011	0.25	0.8	1.35	13.95
2012	0.5	1	2	15.2
2013	1	a	2.75	16.55
2014	1.75	a	3.75	18.15
2015	3	a	5.5	20.5
2016	4.25	a	7.25	22.25
2017	5.5	a	9	24
2018	7	a	11	26
2019	8.5	a	13	28
2020	10.5	a	15	30
2021	13.5	a	18	33
2022	16	a	21	36

a - To be determined by EPA through a future rulemaking - no less than 1.0 billion gallons.
b - To be determined by EPA through a future rulemaking.

5.2.2 California Low Carbon Fuel Standard (LCFS)

California's landmark 2006 legislation, the Global Warming Solutions Act (AB32 - Núñez and Pavley) set a statewide GHG emissions target of 20% below 1990 levels by 2020. One significant early action measure in pursuit of this target is the Low Carbon Fuel Standard (LCFS), established by Governor Schwarzenegger with Executive Order S-01-07 (Schwarzenegger 2007). The LCFS calls for a 10% reduction in the carbon intensity of transportation fuels sold in California by 2020, leading to an annual GHG emissions savings of 20-25 million metric tons CO_2e (Yeh, Sumner et al. 2009). While California's total impact on international trade is small, the LCFS process is being closely followed elsewhere in North America[34] and may become a more broadly implemented approach for GHG emission reduction from transportation (Payne 2008).

The LCFS structure uses a market mechanism to achieve efficient emission reductions, allowing blenders to meet their target carbon intensity on an average basis using blending or emissions trading. Policy makers believe that this model has the potential to incentivize investment and innovation (Farrell and Sperling 2007).

[34] In December of 2009 the governors of 11 Northeast and Mid-Atlantic states signed a Memorandum of Understanding, pledging to create a regional LCFS.

Unlike the other policies discussed here, LCFS is not a biofuel support policy *per se*; it is a transportation emissions reduction policy and is technology neutral. However, biofuels "are expected to play a significant role in the short- to medium-term" (Yeh, Sumner et al. 2009).

Beyond the GHG emission reduction that is the core of the LCFS, no environmental or social sustainability provisions have yet been incorporated into the regulatory language. A study commissioned by the California Air Resource Board (Yeh, Sumner et al. 2009) recommends that sustainability concerns be addressed through a reporting requirement alone, coupled with incentives for voluntary certification.

5.2.3 European Union – RED/FQD

The EU policy context for biofuels is composed primarily of the *Directive for the promotion of the use of energy from renewable sources* (RED) (EC, 2009) and the *Fuel Quality Directive* (FQD) (EC, 1998). The RED seeks to expand the fraction of the energy mix (all energy sources, including biofuels, bioliquids, and electricity) that comes from renewable sources. It sets "a 20% target for the overall share of energy from renewable sources and a 10% target for energy from renewable sources in transport." The FQD, on the other hand, aims (among many other things) explicitly to lower the average carbon intensity of biofuels at least 6% by 2020.

The RED is designed such that the target of 20% renewables inclusion in the total energy mix is EU-wide, with individual targets for member states based upon their "starting point, renewable energy potential, and energy mix" (EC, 2009). The states also have discretion as to the policy mechanisms they employ to achieve their targets. Support could include, but is not restricted to, "investment aid, tax exemptions or reductions, tax refunds, renewable energy obligation support schemes including those using green certificates, and direct price support schemes including feed-in tariffs and premium payments" (EC, 2009).

Recognizing the imperative to mitigate the environmental and social impact of biofuel production, policymakers in the EU have instituted sustainability standards for biofuels incentivized under these two directives. In the interest of regulatory harmonization, they formed a joint committee and have implemented the same sustainability standard for the two directives operational as of June, 2010 (EC, 2009). This system includes both mandatory aspects and reporting requirements.

The standard, which the regulators have described as "the most comprehensive and advanced binding sustainability scheme of its kind anywhere in the world (EC, 2009)" contains a set of mandatory criteria. Member states are responsible for ensuring that these rules are met when biofuels are used for compliance with national targets (for renewable mix under RED or GHG reduction under FQD), or receive financial support or investment aimed at environmental protection. The mandatory sustainability rules are as follows[35]:

A) The greenhouse gas emission saving from the use of biofuels and bioliquids … shall be at least 35% at the outset. On January 1, 2017 this threshold rises to 50%, and one year later rises again to 60% for any biofuel production starting in 2017 or later.

B) Biofuels and bioliquids…shall not be made from raw material obtained from land with high biodiversity value, namely land that had one of the following statuses in or after January 2008:
   a. Primary forest
   b. Areas legally designated for protection of nature or of rare ecosystems or species
   c. Highly biodiverse grasslands

C) Biofuels and bioliquids…shall not be made from raw material obtained from land with high carbon stock, namely land that had one of the following statuses in January 2008:
   a. Wetlands
   b. Continuously forested areas
   c. Peatland

D) Biofuels and bioliquids derived from agricultural raw materials cultivated within the EU must be grown in accordance with EC rule no. 73/2009 which sets out "minimum requirements for good agricultural and environmental practices."

Operators are required to show compliance with these mandatory criteria either by providing data to the relevant national authority, complying with an existing bi- or multi-lateral agreement, or using a recognized voluntary scheme. Voluntary schemes are recognized through an assessment process that benchmarks their criteria and ensures that they "have a strong and auditable documentation management system, and adequate standard of independent auditing of producers" (EC, 2009).

Recognizing that many important impacts of bioenergy are beyond the purview of individual producers, the EC requires that member states collect data and report in

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[35] The European Parliament initially sought to include further criteria addressing, for example, community land tenure and workers' rights, but these were dropped 'as doubts remained whether such fixed social sustainability criteria were in line with the rules of the World Trade Organization" European Parliment (2008). EP seals climate change package. Background Paper from the European Parliament's Press Office. Strasbourg. Other sustainability concerns, including some social impacts as well as effect on water, air, land use, and land degradation are mentioned in articles 22 and 23 of the RED (EC 2009). These are not operationalized, but are instead listed as issues to be monitored and considered for future reporting requirements or potential incorporation into mandatory criteria.

detail on macro-scale effects of their consumption patterns. The Commission is then expected to report to the European Parliament every two years, beginning in 2012 on the impacts of the RED and FQD on social welfare both within the community and in other countries – including food price effects, land rights, land use change, and implementation of ILO conventions.

Finally, the EC has established an innovative approach to incentivizing the expansion of renewable electric transport and biofuels made from "wastes, residues, non-food cellulosic material, and ligno-cellulosic material" (European Communities 2009) in the RED. The renewable portion of any grid electricity devoted to transport will be counted at 2.5 times its energy content and any cellulosic liquid biofuel counted at 2 times its energy content toward the national target of 10% renewables in the transport energy market. The policy also encourages the creation of other incentives at the national level to support diverse and beneficial pathways such as energy from wastes, residues, and algae.

5.3 WTO agreements affecting domestic biofuel policies

5.3.1 General Agreement on Tariffs and Trade (GATT)

The architects of the post-war multilateral trading schemes recognized that while negotiated reductions in tariffs and other border measures were central to their efforts, this would not be sufficient to bring about the truly free trade model that was sought. These efforts could be undermined by internal government policies altering the conditions of competition between domestic and imported products, or among imports from different trading partners. In the case of biofuels, this could include percentage mandates, blend limits, physical property or performance requirements, labeling, and environmental performance requirements, among others. The General Agreement on Tariffs and Trade (GATT), signed in 1947, and then updated through its incorporation at the founding of the WTO in 1994[36] is the primary document governing such policies.

The GATT faces the complex task of maintaining members' ability to achieve legitimate goals through domestic policies, while at the same time ensuring that those policies are not more restrictive than necessary and are not structured so as to benefit domestic products, or those from specific trading partners. Differential treatment, however, is not necessarily inconsistent with WTO rules if such treatment is not discriminatory (Mitchell and Tran 2010). As Howse et al. (2006) point out, "…any standard or regulation is likely to benefit some producers and burden others, an inevitable and innocent effect of the given relative costs of compliance among different producers."

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[36] The WTO officially commenced on January 1, 1995. However, the negotiations leading to its founding took place throughout the Uruguay Round of GATT, culminating in the signing of the Marrakesh Declaration, the Agreement Establishing the World Trade Organization, and most of the Agreements in Annex 1 of the WTO in April of 1994.

5.3.1.1 Articles I and III

A central theme in the language of GATT/WTO and of the jurisprudence that has drawn on that language is the concept of non-discrimination, or "no less favorable" treatment. Article I of GATT states that "any advantage, favour, privilege or immunity granted by any contracting party to any product originating in or destined for any other country shall be accorded immediately and unconditionally to the like product originating in or destined for the territories of all other contracting parties" (GATT 1947). This *Most Favored Nation* statute was designed to prevent governments from providing preferential treatment to certain trading partners over others. Article I is complemented by Article III (*National Treatment)*, which sets out similar provisions preventing preferential treatment to domestic products over imported ones.

The language of articles I and III has two basic elements in common: regulatory treatment should not create an *advantage* for one product over a *like product* from a different jurisdiction. The concept of "advantage" has been interpreted broadly to include, for example, exemptions from duties or fees (Canada-Automobiles), preferred intellectual property protection (EC-Trademarks), and freedom from various legal requirements (Colombia-Ports of Entry) (Mitchell and Tran 2010).

Where treatment is the same, there is "effective equality of opportunities" among products imported from different trading partners (article I) and between domestic and imported products (article III) (Switzer and McMahon 2010). It is important to note that under this language, treatment need not be identical, just no less favorable. Furthermore, this treatment is generally considered to be in violation under article III only if the "less favorable treatment [is] explained by the foreign origin" of the goods in question (WTO 2005)

There can be little doubt that the biofuel policies in question will confer what would be called an advantage for some (particularly lower-carbon) fuels over others. This is, in fact, their intent; the Renewable Energy Directive states in its preamble that in order for environmental goals to be realized, biofuels "meeting [its] criteria must command a price premium compared to those that do not" (EC 2009). However, in order for a policy to be found non-compliant under article I or III, a complainant would need to prove that its fuels were accorded a higher GHG score or worse sustainability performance than domestic fuels or other imports. Furthermore, the complainant would need to show that the less favorable treatment its biofuel products received was due to their national origin.

Some have argued that the sustainability criteria in the RED are aimed at protection of domestic biofuel industries (Erixon 2009). Domestic (EU) and some foreign products certainly perform differently on life cycle GHG and other environmental metrics. This difference is due to varying land use and feedstock cultivation contexts and could result in less favorable treatment of foreign products. As a result, any eventual dispute under GATT would probably hinge upon whether or not biofuels with different GHG or other environmental performance are "like" under WTO law.

5.3.1.2 Like Products

Findings under Articles I and III of GATT as well as other agreements often hinge on whether or not products in question are found to be "like." The established test for the likeness of two products was originated by the 1970 GATT working party report on Border Tax Adjustments, which stated that "the interpretation of the term should be examined on a case-by-case basis…Some criteria were suggested for determining, on a case-by-case basis, whether a product is 'similar': the product's end-uses in a given market; consumers' tastes and habits, which change from country to country; the product's properties, nature and quality" (GATT 1970). This statement has since been formalized, in particular by the Appellate Body (AB) in the Japan-Alcoholic beverages dispute (WTO 1998), which determined that likeness can be established based upon the following four criteria[37]:

i. Physical properties of the products
ii. The extent to which the products are capable of serving the same or similar end uses
iii. The extent to which consumers perceive and treat the products as alternative means to satisfying a demand
iv. International tariff classification of the products

Based on this test, regulations that distinguish between pure fossil fuels and biofuels or blends are unlikely to be found in violation of Articles I and III. These two fuel types are not like on the basis of their physical properties (they differ both chemically and physically), they have different tariff classifications, and they are probably perceived by consumers as different products. It could be argued that biofuels and fossil fuels are like on the basis of end use, but even insofar as this is so,

[37] These criteria are meant to be considered together as a unit. The Appellate Body in the landmark EC-Asbestos WTO dispute stated that, "although each criterion addresses, in principle, a different aspect of the products involved, which should be examined separately, the different criteria are interrelated. For instance, the physical properties of a product shape and limit the end-uses to which the products can be devoted. Consumer perceptions may similarly influence – modify or even render obsolete – traditional uses of the products. Tariff classification clearly reflects the physical properties of a product." They go on to state that "In many cases, the evidence will give conflicting indications, possibly within each of the four criteria. For instance, there may be some evidence of similar physical properties and some evidence of differing physical properties. Or the physical properties may differ completely, yet there may be strong evidence of similar end-uses and a high degree of substitutability of the products from the perspective of the consumer." Their decision held that the panel should examine "…the evidence relating to each of those four criteria and, then, [weigh] all of that evidence, along with any other relevant evidence, in making an overall determination of whether the products at issue could be characterized as 'like'" (WTO 2001).

the criteria taken together indicate that these are not, in the eyes of the WTO, "like products." The question of likeness gets more complicated, however, when applied to a regulation distinguishing among different biofuels on the basis of the feedstock from which they were produced, their life cycle GHG intensities, or other environmental and social parameters.

Of the dispute findings regarding "like" products, perhaps the most relevant to the question of biofuel regulation is that of *Mexico – Tax Measures on Soft Drinks and Other Beverages* (Mexico-Soft Drink), in which the U.S. brought a complaint against Mexico regarding its tax on beverages employing sweeteners other than cane sugar. The panel pointed to the above test in finding that sweeteners, whether high fructose corn syrup, beet sugar, or cane sugar are all like products under WTO law, as they "have 'virtually identical' characteristics in terms of physical properties, end-uses, consumer tastes and habits, and tariff classification" (WTO 2006). This finding makes it clear that the use of different feedstocks in production of an otherwise uniform product – as is the case for different biofuels – is not grounds for finding products unlike in the eyes of the WTO.

A second major element in assessing the likeness of products under contention is the concept of "production or processing methods" (PPM) a long-standing point of contention in GATT and WTO jurisprudence. One well-known and widely criticized case hinging on this "process-product distinction" is *United States – Restrictions on Import of Tuna* (Tuna-Dolphin). This 1991 case was brought by Mexico against the U.S. after the expansion of the Marine Mammal Protection Act banned import or sale of tuna caught with nets that also kill dolphins. The dispute settlement panel in this case found that the ban violated trade laws since it distinguished among products based upon production or processing methods (PPM) rather than anything inherent to the products themselves (GATT 1991).

The Tuna-Dolphin finding and other cases have found that PPMs are not generally a legitimate basis for trade distinction (Doelle 2004). However, it has since become clear that regulations based on environmental PPMs can be compliant or non-compliant depending on their rationale and their implementation (Charnovitz 2002).

The findings in Tuna-Dolphin and Mexico-Soft Drink indicate that different lots of biofuel would probably not be found "like" on the basis of their physical properties or their end uses. While the tariff classification of various biofuels is in flux, the policies in question do not distinguish among different types of fuel, but among *different shipments of the same type of fuel*, differing only in their production specifics meaning their tariff classifications will be the same. This leaves varying consumer perceptions – the extent to which consumers distinguish among fuels on the basis of life cycle GHG intensity or other environmental and social parameters – as the only remaining recourse for maintaining that the fuels subject to the regulations in question are not like.

Consumer preference:

One relevant early case relating to consumer preference is *Spain – Tariff Treatment of Unroasted Coffee* (Spain-Coffee). At issue in this dispute was a Spanish import tariff structure that distinguished among various strains of coffee beans. Spain argued that differences "existed between them, as a result of climatic and growing conditions as well as methods of cultivation." It further claimed that "various types of coffee could not be regarded as 'like products'…[particularly] in the Spanish market where, for historical reasons, consumers' preference for the various types of coffee was well established" (GATT 1981). The case was eventually found against Spain, but is considered a landmark for PPM-based regulation not for what was found, but for what was not found. The Panel did not reject outright Spain's contention that production methods made for unlike products in this case; instead it found that the products were like because the coffees were sold as blends, preventing consumers from distinguishing among them.

A second case that has bearing on the question of consumer preferences is *European Communities – Trade Description of Sardines* (EC-Sardines) brought against the EC by Peru regarding a regulation that prevented Peru and other exporters from using the term "sardines" to describe their products. The dispute settlement panel in this case found the consumer expectations in question were, in effect, *created by* the policy under consideration. The finding states:

> *"If we were to accept that a WTO Member can 'create' consumer expectations and thereafter find justification for the trade-restrictive measure which created those consumer expectations, we would be endorsing the permissibility of 'self-justifying' regulatory trade barriers (WTO 2002)."*

The EC-Sardines finding makes clear that while consumers desire could tip the scales in favor of a biofuel sustainability policy, that desire must not be, in the eyes of the WTO, manipulated by policies under consideration (Switzer and McMahon 2010).

Another important case involving consumer perceptions of likeness is *European Communities – Measures Concerning Asbestos and Products Containing Asbestos* (EC-Asbestos), a dispute brought by Canada against the EC following France's prohibition on import or sale of all products containing asbestos. The initial dispute settlement panel found that asbestos-containing products and their alternatives not containing asbestos were "like" on the basis of their physical properties and uses (WTO 2001). However, an appellate body overturned this decision, pointing to the health risks associated with the use of asbestos and asserting that "…carcinogenicity, or toxicity, constitutes, as we see it, a defining aspect of the physical properties of chrysotile asbestos fibres" (WTO 2001).

This determination of physical difference does not, however, necessarily mean that the products are not "like." The AB held that "in a case such as this, where the fibres are physically very different, a panel cannot conclude that they are 'like products' if it *does not examine* evidence relating to consumers' tastes and habits" (WTO 2001 - emphasis original). The AB pointed out that the consumer of raw asbestos fibers will, in most cases, be a manufacturer of other products "such as cement-based products or brake linings". They go on to state:

> *"…consumers' tastes and habits regarding fibres, even in the case of commercial parties, such as manufacturers, are very likely to be shaped by the health risks associated with a product which is known to be highly carcinogenic. A manufacturer cannot, for instance, ignore the preferences of the ultimate consumer of its products. If the risks posed by a particular product are sufficiently great, the ultimate consumer may simply cease to buy that product. This would, undoubtedly, affect a manufacturer's decisions in the marketplace. Moreover, in the case of products posing risks to human health, we think it likely that manufacturers' decisions will be influenced by other factors, such as the potential civil liability that might flow from marketing products posing a health risk to the ultimate consumer, or the additional costs associated with safety procedures required to use such products in the manufacturing process"* (WTO 2001).

This finding is very significant in that it opened the door to a broader interpretation of "likeness," allowing considerations (in this case, carcinogenicity) other than practical properties of the product. The AB stated that even given this physical difference between the products, they could still be found like given enough evidence in the other criteria, but that there would be a "higher burden" on a complainant to prove their likeness.

The three disputes discussed above are frequently referenced in light of the potential for regulatory distinction based upon differences in consumer perception. In assessing the implications for biofuel sustainability regulations, we recognize an important common thread among these three cases. Notably, the deciding body in each case, while alluding to consumer perception, stopped short of a precedent-setting decision that consumer preference was sufficient to establish that products are not like. In each, the final decision turned on a slightly different consideration. The products were found indistinguishable because of mixing in Spain-Coffee, the perception was based on what was actually a physical difference (carcinogenicity) in EC-Asbestos, and the preference was manipulated by policy in EC-Sardines. This leaves open the question of whether consumer preference for low-GHG or otherwise environmentally sustainable fuels will be found sufficient to justify differential policy treatment.

Assessing "likeness" in biofuel regulations:

Biofuels with different GHG or other environmental and social impacts are nevertheless the same fuel for the purposes of tariff classification, and so would probably be found to be like on that basis. It also seems clear that from the standpoint of end uses, these fuels are "like." The question of physical characteristics is slightly more complex, but the precedent of *Mexico-Soft drink* indicates that they will be like on this basis as well. If there were differential GHG emissions associated with the *combustion* of the fuels, an argument could be made to distinguish among them claiming a physical difference, but the GHGs in question are emitted in production, rendering this argument unviable. Having excluded three of the four criteria for establishing that products are not like, any such argument would necessarily hinge on the question of consumer preference.

As noted above, the Appellate Body in EC-Asbestos held that where products are physically different, there is a 'higher burden' to establish likeness. Mitchell and Tran (2010) assert that the inverse can be inferred, implying that where products are physically like, it will be difficult to prove that they are nevertheless unlike on the basis of consumer preferences. They hold that "it is unlikely that consumer tastes and preferences could be so strong as to displace that presumption, given that the biofuels in question will be physically identical with exactly the same end use capabilities."

Some commentators, however, (Charnovitz, Earley et al. 2008; Switzer and McMahon 2010) hold a different opinion, arguing that consumer preference for more sustainable biofuels – especially if it resulted in concerted lobbying efforts – might be sufficient to establish that different fuels are unlike. Furthermore, even if insufficient today, public sentiment surrounding environmental and social impacts of consumption decisions is developing. This is evidenced by the rapid growth in market share for products such as organic foods, Forest Stewardship Council certified lumber, and various Fair Trade goods. These preferences could become sufficient to justify treating fuels as "unlike" based on their relative GHG performance or other social and environmental impacts.

The question of whether or not biofuels being differentiated in regulation will be found "like" will hinge in part upon how the regulation in question is designed. In order to improve the probability that these fuels will not be found "like," it would be important to ensure that regulatory structures do not require biofuels to be blended with one another before the point of sale. Policies would also need to be designed such that they did not limit, *ex ante*, consumer access to certain types of biofuels (Ackrill and Kay 2010). These design characteristics would allow consumers to express preferences that could then be used to justify the regulatory distinction among the fuels in question.

However, even if designed as described above, most trade experts who have examined the issue in detail believe that biofuels differing only on the basis of life cycle GHG emissions or other upstream environmental and social impacts would be found to be "like" in any eventual WTO dispute (Swinbank 2009; Ackrill and Kay 2010; Mitchell and Tran 2010). It would be extremely difficult to argue that a biofuel with, for example, a life cycle GHG emission savings of 34% is fundamentally different than one with a savings of 35% unless some objective criterion underpins the 35% threshold. This would lead the regulations in question to be found in violation of GATT article I and/or III. This does not, however, necessarily mean that policies distinguishing among such fuels are WTO non-compliant. Precedent suggests that the defense of such a policy would likely resort to GATT article XX, which provides important exceptions to the rule of common treatment for like products.

5.3.1.3 GATT Article XX

GATT article XX lays out a set of "general exceptions" to GATT obligations, describing circumstances under which governments can implement otherwise GATT non-compliant regulations in order to achieve certain public policy goals. Article XX lays out ten such types of measures "that are recognized as exceptions to substantive obligations established in the GATT 1994, because the domestic policies embodied in such measures have been recognized as important and legitimate in character" (WTO 2001). This article provides the most important "textual hooks" for regulating the sustainability of biofuels (de Vera 2007).

Two of the exceptions created by article XX – laid out in paragraphs b and g – are particularly relevant to biofuel sustainability regulations.[38] According to article XX, otherwise non-conforming measures can be taken if:
(b) necessary to protect human, animal or plant life or health
(g) relating to the conservation of exhaustible natural resources if such measures are made effective in conjunction with restrictions on domestic production or consumption

A policy's justification under these exceptions, however, is conditional upon its also fulfilling the requirements of the article's preambular paragraph, or "chapeau." The chapeau requires that measures must not be "applied in a manner which would constitute a means of arbitrary or unjustifiable discrimination between countries where the same conditions prevail, or a disguised restriction on international trade (GATT 1947)."

Article XX(b)

GATT article XX(b) allows policies that are "*necessary to protect human, animal or plant life or health"* In their reading of this article, dispute panels generally assess the word "necessary" separately from the word "protect," first determining whether the product being considered does indeed cause the health effect in question, then

[38] Some biofuel policies might find justification under the "security exceptions" of GATT article XXI: "Nothing in this Agreement shall be construed...to prevent any contracting party from taking any action which it considers necessary for the protection of its essential security interests ... taken in time of war or other emergency in international relations." General Agreement on Tariffs and Trade (1947). Geneva, GATT. This could possibly be used to justify incentives to domestic biofuel industries on the grounds that this action is necessary to build and maintain energy security. This provision would not, however, apply to the biofuel policies being investigated here, as they are primarily aimed at GHG mitigation and broader environmental and social sustainability rather than domestic biofuel expansion *per se*.

assessing whether the policy response in fact protects against those health impacts, and finally evaluating whether the policy was "necessary" to do so (WTO 2001).

The word "necessary" in this text has been read quite literally in WTO disputes to date. In order to justify a policy under article XX(b), nations must demonstrate that there is no other "reasonably available" measure to address the same issue while being "consistent with the General Agreement, or less inconsistent with it" (WTO 2010). This language has been interpreted as explicitly balancing the effect of the policy against its trade restrictiveness, requiring that measures not be disproportionate to the risks they address (Howse, van Bork et al. 2006; Mitchell and Tran 2010).

Article XX(g)

GATT article XX(g) enables creation of policies that are "*relating to the conservation of exhaustible natural resources*." It is worthy of note that where XX(b) requires that policies be "necessary" to address a health concern, XX(g) only requires that a measure be "relating to" the conservation goal. This has been defined as requiring a "real connection" (WTO 1998), or what has been termed a "rational nexus" (Howse, van Bork et al. 2006) between the measure and the protection of exhaustible natural resources, and has led to a much broader interpretation than XX(b). Where the central question in much of the XX(b) jurisprudence has hinged on "necessity," most of the cases looking at XX(g) hinge on the definition of "exhaustible natural resource," and how far governments can go to protect them.

Precedents relating to article XX(g) can shed light on the possibilities for biofuel regulations. One particularly relevant case is *United States – Standards for Reformulated and Conventional Gasoline* (US-Reformulated Gasoline). This 1995 complaint was brought by Brazil and Venezuela against the United States concerning gasoline formulation guidelines under the Clean Air Act. In this dispute, the U.S. justified its regulation under article XX(g), maintaining that clean air is an "exhaustible natural resource…since it could be exhausted by pollutants such as those emitted through the consumption of gasoline." Venezuela countered, claiming that the clean air is "…not an exhaustible natural resource within the meaning of Article XX(g); rather, its 'condition' changed depending on its cleanliness" (WTO 1996). The dispute settlement panel eventually rejected this technicality, finding that the altered condition of the clean air could in this case, constitute a form of consumption. Despite this finding, the panel, and a subsequent appellate body held (for slightly different reasons) that the American regulation was, as implemented, in violation of the General Agreement.

The precedent set in US-Reformulated Gasoline sheds light on the possible disposition of the biofuel policies considered here relative to article XX(g). Though the WTO has not yet weighed in on the issue, it is probable that a stable climate would follow clean air in being found as an exhaustible natural resource. This means that biofuel policies aimed at GHG emission reduction might be provisionally justified under XX(g).

Another critical precedent in understanding article XX(g) is *United States – Import Prohibition of Certain Shrimp and Shrimp Products* (Shrimp-Turtle). This 1996 dispute, brought by India, Malaysia, Pakistan, Thailand, and the Philippines, questioned the legality of a U.S. ban on shrimp caught with equipment that violated U.S. domestic law. Specifically, the U.S. brought import restrictions in line with the Endangered Species Act requirement that domestic shrimping boats be equipped with Turtle Excluder Devices (TEDs) in order to prevent sea turtle by catch.

Since the turtles themselves are not contained in the final product, this regulation was found to distinguish among like products based on PPMs. This also meant that the Convention in International Trade in Endangered Species (CITES) could not be invoked, as there was no trade in the actual endangered species (Sampson, 2001). The dispute settlement panel found that the regulation was not justified under article XX(g) because the turtles were not caught in U.S. waters, so the resource, while exhaustible, was not within U.S. jurisdiction for conservation measures (WTO 1998).

Upon appeal, however, this decision was reversed, in part because the turtles in question were part of a larger, migratory population that was also found in U.S. waters. It is notable that the appellate body in this case still stopped short of truly addressing the question of extraterritorial jurisdiction by implying that its finding was based, at least in part, upon the migratory nature of the turtle population being protected and its existence in US waters (WTO 2001). Still, Shrimp-Turtle is a critical precedent in that it broadened the reading of "natural resources" to include living and ecological resources alongside the mineral resources for which it was intended and to which it had previously been applied (Payne 2008). In doing so, it "brought GATT case law much closer to a reasonable balance between environmental and trade interests" (Chang 2005).

These cases set precedents that may prove important for biofuel regulation. GATT article XX(g) seems to open the door to PPM regulations, but care must be exercised in designing them with enough flexibility to pass the requirements of article XX's chapeau or they will not be compliant.

Chapeau

The preambular paragraph, or "chapeau" of article XX states that the exceptions stated therein all hold, provided that :

> *"…such measures are not applied in a manner which would constitute a means of arbitrary or unjustifiable discrimination between countries where the same conditions prevail, or a disguised restriction on international trade…"*

This language was drafted to prevent abuse by members of the freedoms created by the general exceptions of Article XX (Switzer and McMahon 2010). These stipulations have led policies that would otherwise have been compliant under article XX to be found in violation because of their implementation. A good example of the practical impact and implementation of the chapeau can be found in Shrimp-Turtle.

Although the regulations limiting shrimp imports under the US Endangered Species act were found by the Appellate Body to be provisionally justified under GATT article XX(g), the implementation of the policy was found to create an "arbitrary and unjustifiable discrimination" and thus to violate the terms of the *chapeau* (WTO 2001). At issue was not whether sea turtles should be conserved; in fact each of the complaining nations had its own sea-turtle conservation program. The question was whether conditioning access upon the adoption of a *specific* conservation measures was permissible under WTO rules. It was found that such inflexible, unilateral action constituted arbitrary discrimination.

Recognizing that it was the implementation of the rule rather than its substance that was in violation of GATT, the US endeavored to revise its regulation to bring it into compliance. In doing so, they consulted with the nations involved, and developed a statute requiring only that states have regulations that are "comparable in effect" to requirement of TEDs. In allowing states to achieve the (legitimate) conservation goal through whatever means are most appropriate to their own circumstances, the more flexible statute was found to meet the *chapeau* in evoking article XX(g).

The appellate body in Shrimp-Turtle explicitly allowed for unilateral application of conservation measures, stating that "[It] is not necessary to assume that requiring from exporting countries compliance with, or adoption of, certain policies prescribed by the importing country, renders a measure *a priori* incapable of justification under Article XX. Such an interpretation renders most, if not all, of the specific exceptions of Article XX inutile, a result abhorrent to the principles of interpretation we are bound to apply" (WTO 2001). However, the Shrimp-Turtle precedent also indicates that implementation of such policies must be flexible, and that international consultation will make policies much more likely to be found compliant with the requirements of the chapeau (Switzer and McMahon 2010).

The other case cited above – US-Reformulated Gasoline – while provisionally justified under XX(g) was also found in violation of the chapeau. In this case, the violation was due to the finding that there were other ways to apply the Clean Air Act to achieve the same policy outcome while creating less trade distortion.

Implications for biofuel sustainability regulations

While there are no discernible differences in health effect resulting from the use phase of different shipments of a "like" biofuel, governments might be able to use article XX(b) to justify a life cycle GHG-based biofuel policy on the basis of the health effects of climate change. A clear connection exists between climate change and human health (McMichael, Woodruff et al. 2006), and a regulating country could argue that even when emissions occur overseas the health effect is borne domestically owing to the global nature of climate change. A dispute panel would probably find that evidence of a health effect is sufficient, and that the mitigation measure would protect against that effect. However, a life cycle GHG-based biofuel policy would probably not be seen as "necessary" to achieve that goal, as there are numerous other measures that could reduce total GHG emissions, many of them less trade distorting. This fact makes it unlikely that a policy regulating life cycle GHG emissions of biofuels would be justified on the basis of article XX(b).

A policy addressing sustainability criteria other than GHG emissions would be even less likely to be justified under article XX(b). Most of the criteria in question would seek to manage PPM impacts. While international, upstream GHG emissions might be shown to have a domestic health effect, it would be hard to convincingly say the same for upstream water pollution, soil erosion, biodiversity loss, or labor abuses.

Policies regulating life cycle GHG emissions for biofuels would find strongest grounds for justification under the WTO through GATT Article XX(g). Stable climate could be found to be an exhaustible natural resource following the logic used in US-Reformulated Gasoline.[39] Furthermore, that resource is certainly shared, allowing for the extraterritorial jurisdiction necessary to regulate on the basis of PPM emissions.

Policies regulating for non-GHG environmental and social impacts will be more difficult – though not impossible – to justify under article XX(g). Existing precedent in the WTO has left open the question of whether legitimate environmental goals can be addressed by regulations when the impact in question is truly outside the regulating country (as would be the case for most of the environmental concerns addressed by the policies in question). This remains an open question in WTO jurisprudence, as the AB in the Shrimp-Turtle decision expressly declined to rule on whether the natural resource being protected (i.e. migratory turtle populations) *must* be shared to establish regulatory jurisdiction (WTO 2001).

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[39] Alternatively, some commentators have suggested a very direct, though perhaps unlikely, justification for biofuel incentive policies under XX(g); these policies could be framed as protecting the exhaustible *fossil fuel* resource by reducing its use (Ackrill and Kay, 2010).

Table 5-3: Potential for the biofuel policy types considered here to find justification under GATT article XX(b) and (g)

		Biofuel policy in question	
		Life cycle GHG emission	Other sustainability criteria
GATT Article	XX(b)	**Difficult** to justify. Measure is not "necessary" given that other policies could achieve similar emission reductions with less trade restriction.	**Very difficult** to justify. PPM-based policy. Health effects will occur in other countries, making them outside regulatory jurisdiction for XX(b).
GATT Article	XX(g)	**Possible** to justify. Stable climate is probably an "exhaustible natural resource." XX(g) does not require that measure be "necessary," only "relating" to conservation.	**Difficult** to justify. PPM-based policy. Certain criteria may be possible – especially insofar as they are related to GHG mitigation.

The primary challenge in justifying life cycle GHG-based biofuel policies under article XX(g) would lie in whether the policies were designed so as to meet the strictures of the chapeau. WTO dispute settlement panels have repeatedly indicated that at issue here is whether or not the 'design, architecture, and revealing structure' of the measure under consideration reveals an intent to "conceal pursuit of trade restrictive measures" (WTO 2001; 2001A). The threshold level chosen for inclusion in the Renewable Energy Directive could prove difficult in this regard, as the calculated GHG intensity of European rapeseed biodiesel under the standard is 38% below the petroleum baseline, or just above the threshold of 35%. Some have pointed to this value as evidence that RED is a disguised protectionist measure, designed to give European fuels a competitive advantage over imports, particularly of palm oil biodiesel from Southeast Asia, which has an assigned default value of 19% below the baseline (Erixon 2009; Swinbank 2009).

Precedents suggest several policy characteristics that will be required in order to pass this test. The first consideration is scientific defensibility. Policies, such as RED, LCFS, RFS, and others that seek to mitigate the direct life cycle GHG intensity of biofuels would probably qualify here. Climate science is sufficiently established for GHG intensity to be a basis for a regulatory distinction (Ackrill and Kay 2010). The uncertainties associated with Life Cycle Assessment (LCA) methodologies could be used to call these policies into question, but will probably not provide grounds for their being found non-compliant. The issue raised above, of setting the GHG threshold in a manner arguably motivated by protectionism could prove problematic to regulators, and should be considered in future rule making. On this issue, the LCFS, which allows blending to meet the target could be better positioned than RFS or RED, as its GHG reduction level is set on a blend average basis, thereby avoiding entirely the creation of a hard threshold for inclusion.

Beyond the scientific basis, the next question will be whether environmental impact is the primary goal of the policy in question. Shrimp-Turtle stated that measures must not be "disproportionately wide in [their] scope and reach in relation to the policy objective" (WTO 2001). This means that fuel life cycle GHG and sustainability policies must be written explicitly to address these goals (Ackrill and Kay 2010). The LCFS is a good example of this, as the entire intent of that policy is to reduce the GHG intensity of transportation fuels. Similarly, the RED preamble suggests that the goals of the policy are preservation of certain environments and reduction of carbon emissions to combat climate change (EC 2009).

Once it is established that a policy is primarily aimed at conservation, a dispute panel would then assess whether the same goal could have been achieved in a less-trade-restrictive manner. If biofuel policies are framed as aiming to reduce the GHG intensity of transportation fuels (rather than transportation as a whole, or the entire economy) they will probably stand up to this analysis. Allowing for tradable carbon credits would increase flexibility and minimize distortion, improving the standing of these policies in this regard. California's LCFS model could be found less trade restrictive than, for example, the EU's RED because it regulates fuels on a blend average basis rather than capping the GHG intensity of individual fuels.

The final test under the Chapeau relates to how the policy in question is implemented. The Shrimp-Turtle findings indicate that policies must be flexible and responsive to conditions in various countries. The EU has avoided this issue entirely by leaving implementation of RED at the discretion of the individual member states. This does not, however, exempt the members from WTO dispute for actions taken for RED compliance. The best way to ensure that policy implementation will be found to comply with the chapeau is to engage in meaningful negotiations with supplier nations in the policy making process. This type of multilateral dialogue has not occurred sufficiently in any of the jurisdictions currently promulgating biofuel policies considered here (Swinbank 2009).

5.3.2 Agreement on Technical Barriers to Trade (TBT)

The Technical Barriers to Trade (TBT) Agreement dates to 1994 and is intended to prevent disguised protectionism by balancing the trade principles of GATT with the need to pursue legitimate national goals through the use of technical regulation (Ackrill and Kay 2010). The Agreement applies to a wide range of regulation types, governing the use of technical regulations, defined as any "document which lays down product characteristics or their related processes and production methods…with which compliance is mandatory (WTO 1994)."

TBT supplements the non-discrimination obligations present in GATT. In particular, it requires that regulations be no more trade restrictive than is necessary to meet any of a prescribed list of *legitimate objectives* – a list that largely mirrors the exceptions laid out in GATT article XX. The TBT agreement also stipulates that standards should, wherever possible, be based on "performance rather than design or descriptive characteristics" (WTO 1994). This leaves life cycle GHG-based biofuel policies in good standing, as inclusion is conditioned only on overall performance rather than on any production practice in particular (Payne 2008).

The question of how far upstream in the product life cycle the TBT agreement applies, however, remains an open question. No case has yet been found under this agreement citing PPMs as a relevant product characteristic, and some governments have argued that such policies do not fall under the aegis of the TBT (UNCTAD 2008; Mitchell and Tran 2010).

Biofuel regulations are being promulgated in the face of a changing scientific landscape, as researchers continue to investigate the GHG, land use, and other environmental implications of this burgeoning industry. The TBT agreement leaves the door open to this type of evolutionary policy process, recognizing that policy makers can only account for "*available* scientific and technical information" (WTO 1994).

Finally, the TBT agreement encourages members to base their technical regulations on international standards wherever possible, reasoning that these standards will ease regulatory harmonization and that they represent the least trade-restrictive approach to achieving relevant objectives (Howse, van Bork et al. 2006). The international consultation that goes into such standards also reduces the likelihood that an affected party would bring a dispute (Ackrill and Kay 2010)

5.3.2.1 Harmonization through standards

The TBT agreement lays out the following definition of a standard:

> *"[A] document approved by a recognized body, that provides, for common and repeated use, rules, guidelines or characteristics for products or related processes and production methods, with which compliance is not mandatory (WTO 1994)."*

Importantly, this definition distinguishes standards from technical regulations and from multilateral treaties, with which compliance *is* mandatory. In order to qualify for the purposes of the TBT, a standard-setting organization must be open to all WTO members, though not all members must participate or agree to the standard (Charnovitz, Earley et al. 2008). The TBT further establishes a code of good practice, laying out guidelines for the activities of international standard-setting organizations, but leaves the oversight and management of these entities to the International Organization for Standardization (ISO) and the International Electrotechnical Commission (IEC).

These standards give governments the opportunity to decide whether, and how, to convert the rules therein into mandatory regulations within their own jurisdiction. Provided the above criteria are met, policies based on international standards, are unlikely to be considered "unnecessary obstacle[s] to trade" under the TBT agreement (Charnovitz, Earley et al. 2008).

The international standard-setting process described above has other benefits beyond aiding compliance with the TBT agreement. Given that governments seek to create policies regulating, for example, the life cycle GHG intensity of biofuels, this international process prevents a profusion of different standards, which would impose added costs on producers and create consumer confusion. However, competition and experimentation among standards also has advantages in improving criteria development and certification systems (Charnovitz, Earley et al. 2008). Three major standardization processes for sustainable biofuels have entered the space and are in various stages of development and implementation. These are:

1. *The Global Bioenergy Partnership (GBEP)* – a network of major biofuel producer and consumer nations along with several UN agencies. GBEP's mandate is to coordinate research, development, and policy frameworks surrounding biofuels (Van Dam, Junginger et al. 2007). It has developed a set of 24 sustainability indicators.
2. *The International Energy Agency (IEA) Bioenergy Task 40* – convened under the auspices of the IEA, this initiative is devoted to trade in sustainable biofuels, acting as a knowledge broker on important and contentious issues such as land use change.
3. *The Roundtable on Sustainable Biofuels (RSB)* – a multi-stakeholder initiative, using private certification to improve the environmental performance of biofuels. The RSB standard was developed through cooperation with a range of stakeholder types including NGOs, agribusinesses, governments, UN agencies, and energy companies.

5.3.3 Agreement on Sanitary and Phytosanitary Measures (SPS)

The agreement on Sanitary and Phytosanitary Measures (SPS) was designed to allow states to enact measures protecting their territory from certain types of risks that may stem from some international trade. Policies covered by the SPS include conditions on agricultural imports to protect against "risks arising from the entry, establishment or spread of pests, diseases, disease-carrying organisms or disease-causing organisms" (WTO 1994). The SPS grants states significant leeway in protecting against these risks, but mandates that they use the least trade-restrictive measures available to achieve their goals and that international standards be applied where possible. Finally, unlike TBT, SPS requires that policies be based on scientific evidence and risk analysis (Howse, van Bork et al. 2006).

It is unlikely that SPS will be the basis of any trade dispute regarding the policy types considered here, since the agreement could only apply to trade in biofuel feedstocks, not in the fuels themselves, which pose no phytosanitary risks. Furthermore, SPS is only brought to bear on policies that address *domestic* risks, meaning that the agreement does not apply to the supply chain impacts that are the subject of most biofuel regulation (Charnovitz, Earley et al. 2008).

5.3.4 Agreements governing subsidies

Many of the most significant policies affecting the biofuels market are subsidy-related, as governments have supported nascent biofuel industries through border protection and domestic subsidies (Doornbosch and Steenblik 2007). In the EU, for example, domestic biofuel production is being promoted under the RED with tax exemptions, feedstock production subsidies, and market price supports adding up on average to €0.44 per liter petroleum equivalent for ethanol and €0.30 for biodiesel. Some EU member states are providing much more support. The ethanol excise tax exemption in Germany, for example, is €0.65 per liter (Jung, Dörrenberg et al. 2010). In the United States, ethanol blenders receive a $0.45/gal excise tax credit, rising to $0.55/gal for small ethanol producers, and $1.01/gal for producers of cellulosic fuels. The protective nature of this subsidy is further compounded by the $0.54/gal tariff applied to all imported ethanol. For biodiesel, the tax credit is $1.00/gal.

One of the main functions of the WTO is the provision of mechanisms for the negotiated reduction of tariffs and subsidies. Members commit to a transparent "bound" tariff rate for each class of goods and cannot charge tariff in excess of this rate to goods from any other member. They can, however, apply tariffs lower than the bound rate, either to all trading partners or to certain members as part of a negotiated free trade agreement.

Central to the WTO disciplines on subsidies is the classification of products. Most members use the Harmonized Commodity Description and Coding System (HS) developed and used by the World Customs Organization. To date, ethanol has been classed as an agricultural good, while biodiesel is classed as an industrial good, subjecting the two types of fuels to very different treatment under the WTO as they fall under different treaties.

Some stakeholders advocate the creation of a new HS code for renewable energy products as a solution to this classification problem (Switzer and McMahon 2010). Others propose avoiding this inconsistency, and allowing biofuel-specific negotiations by classifying these fuels for WTO purposes as an "environmental good" (Christian 2009). Howse (2006), however, points to the difficulty in defining "environmental good," arguing that "linking biofuel trade liberalization to a specific set of environmental goals would be a mistake. Environmentalists are themselves divided on whether particular biofuels are an overall positive for the environment…The WTO is not a desirable forum for resolving such complexities."

Members' general obligations under GATT are augmented in the WTO through incorporation of mechanisms intended to address specific issues. Three of these disciplines have implications for the subsidy options available to nations for supporting biofuel programs. These are the Agreement on Subsidies and Countervailing Measures (SCM), the Agreement on Agriculture (AoA), and the Generalized System of Preferences (GSP). The primary focus of this paper, however, is on standards and regulations addressing the environmental performance of biofuels rather than subsidies devoted to expanding their market share. For this reason, a more detailed treatment of the above WTO disciplines, and their implications for biofuel subsidies can be found in annex 1 to this chapter.

5.4 Conclusions

The most certain way to resolve the issue of WTO compliance for biofuel policies is to avoid a dispute altogether. There are countless policies currently in operation across the globe that technically contravene WTO obligations, but against which no dispute has been brought. This can be due to a number of factors. Often, violations have a small enough impact on trade that the injured nation cannot justify the expense of a dispute. Furthermore, political economy is perhaps the largest driver in determining which disputes are pursued in the WTO; small nations often have too much to lose and too little to gain from bringing a dispute against their much larger and more powerful trading partners.

RED in the EU and RFS in the US both reduce the risk of WTO dispute by not prohibiting high-GHG or otherwise unsustainable fuels outright, but instead establishing conditions for access to benefits. This is a less trade-restrictive approach (Mitchell and Tran 2010). On the other hand, California's LCFS covers *all fuels* sold in the state and thereby opens the policy to some WTO dispute risk. In other ways, however, the LCFS structure is *more* compliant with WTO disciplines than its US and EU counterparts in that it allows the GHG requirement to be met through blending. This eliminates the need to establish an arbitrary threshold for exclusion and avoids barring any specific fuels outright – both potential violations of Articles I and III of GATT.

Regulators in the EU seem to have been particularly cognizant of WTO obligations when developing their sustainability criteria. A great deal of outside input – including from major trading partners – was solicited throughout the process, which also drew extensively from negotiated international agreements (Ackrill and Kay 2010). Also, the criteria themselves apply to all biofuels equally, and draw a sharp distinction between mandatory criteria and reporting requirements. Finally, the fourth sustainability criterion, dealing with general agricultural best practices, many of which would not be WTO compliant, only applies to biomass produced *within* the EU.

If a dispute is brought against RED, RFS, or LCFS – and there is evidence that some will be – their compliance with GATT will be the central issue. Each of these policies would probably be found *prima facie* in violation of GATT Article I and/or III based on their differential treatment of biofuels based on life cycle GHG emissions or sustainability criteria. It is possible that these fuels would not be found to be "like" owing to consumer preferences for low-carbon or otherwise sustainable fuels, though such an outcome is improbable judging from precedent. As a result, regulators would probably look to article XX to find justification for these policies.

The precedents discussed in this chapter signal the probable standing of life cycle GHG-based biofuel regulations in light of GATT Article XX. Given the global nature of climate change and the fact that GHG emissions all over the world are essentially identical in their effect, policies distinguishing among biofuels on the basis of their life cycle GHG footprint might be provisionally admissible under GATT article XX(g). Such policies may also be permitted under article XX(b) insofar as climate change can be tied to human health effects.

These policies *may*, however, run into difficulty insofar as they use GHG intensity to create a firm threshold for inclusion. These hard lines, such as the 35% GHG reduction threshold for inclusion under RED, could be considered more trade restrictive than necessary under the article XX chapeau or under TBT (Mitchell and Tran 2010). For this purpose, LCFS might be better positioned, as its blend average model creates incentives for improvement along the GHG-intensity spectrum without banning fuels outright.

Policies regulating for broader environmental and social sustainability would not be likely to find similar justification if brought under dispute. Most of the criteria of concern, from soil erosion to water pollution to various social implications cannot similarly be said to represent shared exhaustible natural resources, and so will not probably fall within the jurisdiction of the regulating nations. Ackrill and Kay (2010) assert that RED does not contain mandatory social criteria because "it was felt that such rules would step over some peoples' red lines and thus would almost certainly trigger an action. A successful defense could not be guaranteed and, moreover, such an action could threaten the entire structure of sustainability criteria."

An exception to this assessment would apply for those broader sustainability criteria that can be explicitly linked to GHG emissions and so can find their justification in the WTO on the basis of climate mitigation. For example, the EU's stipulation that fuels counted towards RED requirements must not be produced on high GHG value lands, serves to protect key ecosystems such as forests, wetlands, and peatlands. These habitats are worthy of conservation in their own right, but casting their protection in the language of climate mitigation increases the likelihood of that protection being found WTO compliant. The WTO standing is more uncertain for the corresponding requirement under RED to protect high biodiversity value lands.

As we can see from Shrimp-Turtle, the permissibility of protective measures, even those that are provisionally justified under article XX, depends upon their implementation. The final US shrimp import regulation laid out standards that must be met, while leaving flexibility for the regulatory regimes that could be implemented in meeting them. A biofuel policy could be designed in a similar manner – creating a variety of avenues through which producers could become certified. RED avoids this issue by leaving the creation of specific policies to fulfill its targets up to the individual member nations. This does not, however, protect these members from disciplinary action if their resulting policies do not comply with WTO regulations.

Despite arguments presented above, it is entirely possible that even the GHG-based biofuel policies would be found not to comply with WTO obligations because of their inherent PPM nature or because these policies will inevitably "discriminate" against trading partners whose fuels have a larger climate footprint. If this were to occur, the WTO *itself* would be in a very difficult position, as this would indicate that the orthodox interpretation of its agreements is unable to adapt to the present, unique challenges of climate change.

As de Vera (2007) observes, "[t]hough the WTO must continue to be vigilant against protectionist measures made in the name of environmental preservation, rulings that give short shrift to genuine efforts aimed at preventing environmental catastrophe potentially undermine the WTO's legitimacy." This predicament could lead to a shift in interpretation of WTO agreements and obligations to be more permissive of legitimate environmental conservation measures. For example, the prohibition of PPM-based environmental regulations has been called "increasingly out of sync with market realities" (Araya 2003) and may begin to soften. Similarly, the interpretation of "exhaustible natural resource" in article XX(g) could continue its recent shift (Shrimp-Turtle, US-Reformulated Gasoline) towards encompassing more of the sustainability concerns at issue in the biofuel space. As the AB in Shrimp-Turtle made clear, the meaning of natural resource is "by nature evolutionary," as it is based on "contemporary concerns of the community of nations about the protection and conservation of the environment" (WTO 2001). Finally, the unique circumstances of climate change mitigation could also extend to the application of the article XX chapeau, which the Appellate Body in Brazil-Tyres stated is not "fixed and unchanging" but rather "moves as the kind and shape of the measures at stake vary" (WTO 2007).

Still unknown, however, are the trade law implications of biofuel policies designed to address the burgeoning concern of indirect land use change (iLUC) (Searchinger, Heimlich et al. 2008; Melillo, Reilly et al. 2009; Al-Riffai, Dimaranan et al. 2010; Hertel, Golub et al. 2010). There remains little question that iLUC is a real effect and that life cycle GHG accounting for biofuel production is incomplete without its being included in some way. However, these policies go a step beyond PPM in managing effects that are mediated by international commodity markets. While there is no directly relevant precedent, it is possible that these policies would be found in violation of WTO law on the basis of scientific defensibility, given that quantification of this effect depends on a very new and shifting body of research.

The evidence investigated in this chapter suggests that at least some important sustainability parameters would be unlikely to hold up under WTO scrutiny. To manage these impacts, governments may need to use mechanisms other than regulation. In the words of a WTO ministerial group: "difficulties, the origins of which lie outside the trade field cannot be redressed through measures taken in the trade field alone" (WTO 1993). In order to avoid some of the negative environmental and social effects that have been associated with biofuels, multilateral negotiations will need to be undertaken.

Through a multilateral environmental agreement, governments can choose to enact policies that are in the social best interest but would otherwise be in violation of WTO obligations. An example of this is the Kimberly Process Certification Scheme (KPCS) for preventing so-called conflict or "blood" diamonds from entering mainstream markets. This negotiation codified what would otherwise be prohibited PPM-based regulations if imposed unilaterally. The KPCS was taken one step further in 2006, when it obtained a waiver allowing its signatories to enforce its conditions on non-signatories (Charnovitz, Earley et al. 2008).

A multilateral environmental agreement does not need to operate in isolation and can instead expand the reach, relevance, and adaptability of existing trade structures. The global climate agreements, for example, borrow extensively from GATT in ensuring that policies it enacts cause minimal trade distortion and that compliance is not used as a cover for disguised protectionism. A similar structure could be used for biofuel sustainability, whether under the auspices of the existing climate convention or as a free-standing negotiated agreement. Such a negotiation would find a strong foundation in existing international initiatives such as the Global Bioenergy Partnership, the IEA Bioenergy Task 40, and The Roundtable on Sustainable Biofuels.

5.5 ***Annex 1*** – WTO agreements governing subsidies

5.5.1 Agreement on Subsidies and Countervailing Measures (SCM)

The agreement on Subsidies and Countervailing Measures[40] (SCM) governs specifically the subsidies that WTO member nations can apply to goods in their jurisdiction. A subsidy, as defined in the SCM either has a cost to the government (i.e. direct cash payments or tax exemption) *or* "there any form of income or price support" and "a benefit is thereby conferred" (WTO 1994).

Two forms of overtly protectionist subsidy are prohibited outright – those attached specifically to exports and those "contingent…upon use of domestic over imported goods" (WTO 1994). Other types of subsidies, though not prohibited, can be "actionable" if they have certain adverse trade effects (Howse, van Bork et al. 2006). To be actionable, support must conform to the definition of subsidy laid out above, and must be proven to have an adverse effect on other WTO members' domestic industries. When a subsidy is found to be "actionable," members are within their

[40] A "countervailing measure," under the SCM agreement, is an import duty imposed unilaterally with the intent of neutralizing a competitive advantage enjoyed by imported goods over their domestic analogs resulting from prohibited subsidies in their country of origin.

rights to impose a unilateral countervailing measure to correct the market disadvantage.

An example of this from the biofuel industry can be found in the $1/gallon biodiesel blender's tax credit created to incentivize biodiesel production and consumption in the United States. This credit greatly reduces the domestic price of biodiesel. It also, however, reduced prices for exports, and created a loophole known as "splash-and-dash." Taking advantage of this loophole, biodiesel made outside the United States and destined for European markets was brought to US ports and blended with a "splash" of petroleum. This blending entitled the operator to $1/gallon tax credit, making the biodiesel much more competitive in European markets, and functionally subsidizing European fuel with US tax dollars. The US closed this loophole once it was recognized, but not before the EU, under pressure from its biodiesel industry, threatened to impose a countervailing duty of $1 per gallon of fuel imported from the US. This duty would have been justified under the SCM, as it would have corrected the market disadvantage in Europe resulting from the US subsidy.

Some of the subsidies and other support mechanisms at play in biofuel markets *are* allowed under the SCM agreement. For example, a government could subsidize purchases of flex-fuel vehicles to expand the market for biofuels as this would not alter the relative competitive position of domestic vs. imported fuels (Howse, van Bork et al. 2006). Tax incentives to domestic biofuel producers would seem to violate the SCM amounting to a contribution from the government in the form of foregone revenue. However, in order determine the presence of a contribution, a panel would need to establish an appropriate benchmark. Insofar as the biofuel industry is created *by* government action, a usable benchmark may not be present (Switzer and McMahon 2010).

5.5.2 Agreement on Agriculture (AoA)

The second critical WTO discipline affecting biofuel support policies is the Agreement on Agriculture (AoA). Under the AoA, an aggregate measure of support (AMS) is set for each member nation, reflecting the *total amount* of qualifying support that nation can provide to its agricultural sector, and leaving the distribution of that support to the discretion of that nation's government. This AMS is then subject to reduction over time, slowly phasing out agricultural subsidies globally.

Under the AoA, subsidies are grouped into the following categories, or "boxes":

- **Green Box** subsidies have "no, or at most minimal, distortive effect on trade or production," and are therefore not capped under the AoA.
- **Blue Box** subsidies are in a category between the green and amber boxes. These are subsidies of the type that would normally fall into the amber box, but which are attached to production-limiting programs, thereby reducing the international trade distortion of the subsidy program.

- **Amber Box** subsidies are trade-distorting programs, such as price supports, that do not qualify for inclusion in the green or blue boxes, and are therefore covered under the AMS.[41]

Some subsidies for biofuels might be exempted from inclusion under the AMS by being classed in the green box based on environmental benefit. This would apply where governments, for example, mandate a certain biofuel mix or certain production practices in order to achieve environmental goals, and then subsidize the mandated production. In this case, the subsidy could be classified as green box provided the amount of the payment does not exceed the total added cost to producers of complying with the government program (Howse, van Bork et al. 2006).

Finally, domestic support programs can be exempted from inclusion under the AMS by qualifying as *de minimis*, meaning that the support accounts for less than 5% of the total product value. The threshold for recourse to this exception is raised to 10% in less developed countries. In the case of biofuels – especially second generation fuels – *de minimis* exemption for reduction commitments may prove difficult to determine, as biomass is not yet traded as a commodity, leaving its value in flux (Christian 2009).

The AoA will only apply to those biofuels that are classified under the Harmonized System as agricultural goods – only ethanol at present. It would also, of course, apply to any cultivated biofuel feedstocks traded in their raw state. If WTO members determine that it is in the global best interest to allow further subsidy to these goods, they could explicitly exclude relevant products from the AoA by listing them in the Annex to the agreement (Ackrill and Kay 2010).

5.5.3 Generalized System of Preferences (GSP)

The WTO agreement also sets out the Generalized System of Preferences (GSP) – a structure enabling governments to adopt differential and more favorable tariff rates for some trading partners under certain circumstances. Such preferential treatment must be non-reciprocal and can be adopted, for example, to address "development needs" or sustainability concerns in LDCs (Howse, van Bork et al. 2006). This allowance for preferential treatment stems, at least in part from the "common but differentiated responsibilities" recognized under the Rio Declaration owing to the "responsibility that [developed countries] bear in the international pursuit of sustainable development in view of the pressures their societies place on the global environment" (UNCED 1992).

Unlike the bound tariff rate, treatment under the GSP can come with conditions that would otherwise be in violation of the WTO agreement. For example, during the

[41] A more detailed treatment of the Agreement on Agriculture and its implications for biofuel support policies can be found in Switzer and McMahon (2010)

Cold War, the US denied GSP status to communist nations (de Vera 2007). At other times, specialized tariff treatment was given by the EU to certain countries as part of an incentive program to combat drug trafficking.

In the biofuels case, some ethanol exporters are already benefiting from EU and US trade preference under the GSP (Oosterveer and Mol 2010). The conditionalities allowed under the GSP could be used to incentivize environmentally sustainable production practices in exchange for the preferential tariff rates. A comparable model is already in place for timber imports to the EU, which provides tariff preferences only to timber certified by the International Timber Trade Organization (de Vera 2007).

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