12 | Qualitative Analysis of Environmental Effects of Algae Production

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12.1 Introduction

This chapter addresses the environmental effects of potential algal biomass production for biofuels and bioproducts, as described in volume 1 of the *2016 Billion-Ton Report* (*BT16*) (DOE 2016). The chapter emphasizes greenhouse gas (GHG) emissions and water consumption, and considers effects of potential algal biomass production on other environmental indicators. The scenarios include algae production that is co-located with waste CO_2 sources in the conterminous United States.

Microalgae and cyanobacteria are widespread and highly efficient photosynthetic organisms that can use sunlight and nutrients (carbon dioxide [CO_2], nitrogen, phosphorus, and trace metals) to create biomass. Algal biomass contains lipids, proteins, and carbohydrates that can be converted and upgraded to a variety of biogas and biofuel end products, including but not limited to hydrogen, methane, renewable diesel, biodiesel, aviation kerosene, gasoline, butanol, and ethanol. (In pathways not considered in *BT16* volume 1, ethanol can be produced directly by organisms that serve as biological catalysts.) Bioproducts derived from algae include livestock feed, nutritional supplements, and plastics.

Unlike the terrestrial biomass described in earlier chapters, algal biomass for biofuels is not yet economically viable, despite the potential benefits of high biomass yields per unit area and significantly higher energy content per unit mass compared to other terrestrial bioenergy feedstocks (Singh et al. 2011). The smaller-scale production of algae for high-value bioproducts, such as nutritional supplements, fertilizers, and cosmetics, is already economically viable. Technological advances are needed to make algae for biofuel cost-competitive. Because the energy-scale production of algae, especially for fuels, has not yet been demonstrated (White and Ryan 2015), environmental effects of commercial-scale cultivation systems have also rarely been investigated in the field.

The objective of this chapter is to provide a qualitative analysis of environmental effects of the algal biomass potential estimated in *BT16* volume 1. In contrast to the other analyses in this report that focus on three specific price scenarios in 2017 and 2040, this chapter considers aspects of many algal biomass supply and price scenarios from volume 1.

12.2 Scenarios

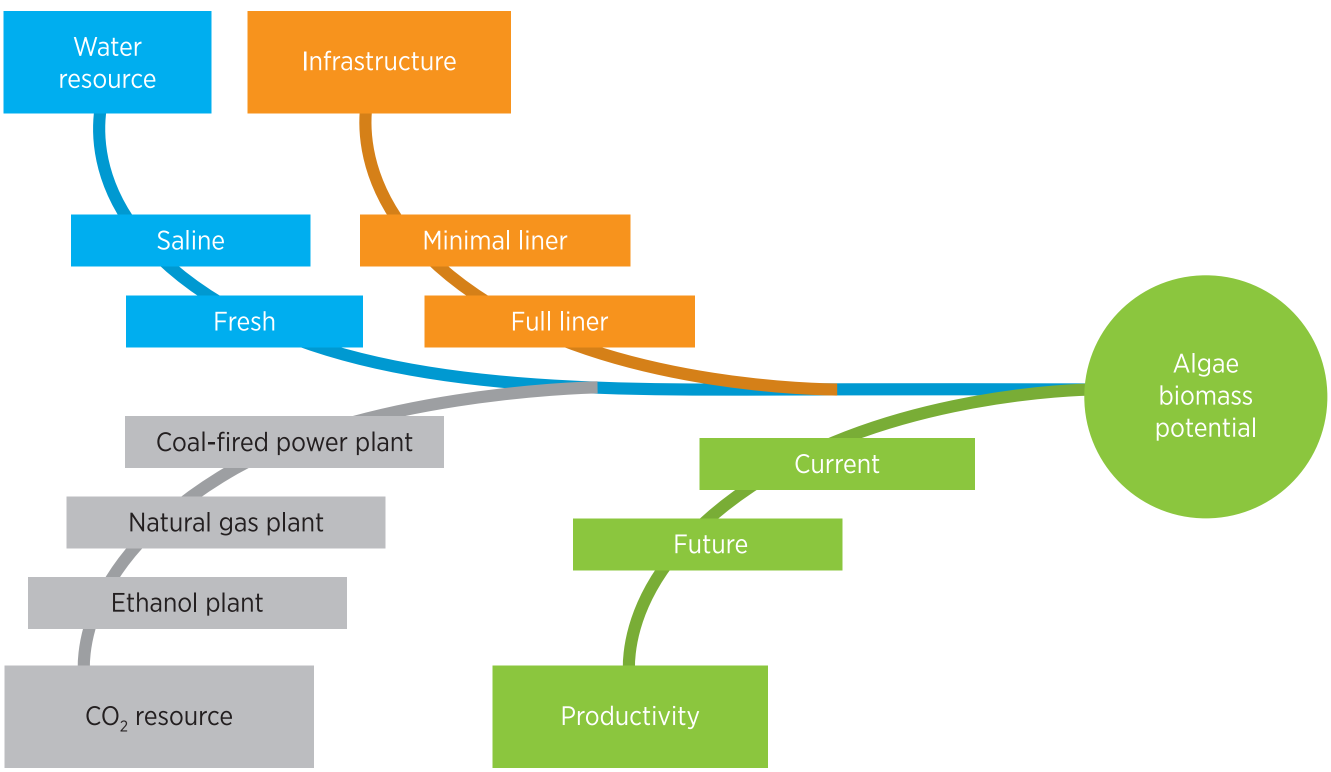
The scenarios from *BT16* volume 1 comprise a subset of the algae production potential that could be co-located with CO_2 sources, i.e., ethanol-production plants, coal-fired power plants, or natural gas–fired power plants. The potential algal biomass represents cultivation at distances from CO_2 sources that would represent cost savings compared to the commercial purchase of CO_2. CO_2 co-location is a strategy used in *BT16* volume 1 to quantify the most likely locations and quantities of algal biomass production in lieu of the strategy used to identify economically available agricultural biomass, i.e., modeling the economics of land management alternatives and selecting the most profitable option for each county. Costs for algae cultivation were taken from an established techno-economic model and recent DOE production design case study (Davis et al. 2016).

The variables that were combined to define the scenarios in the algae analysis in *BT16* volume 1 are depicted in figure 12.1, and rather than providing abbreviations for the scenarios, this chapter and *BT16* volume 1 refer to scenarios as combinations of variables. Potential algal biomass production was estimated for algae grown in open pond-raceway systems that included 405 hectares (ha)—1,000 acres—of pond area. Ponds were ten acres in size and 30 cm in depth, with 100 ponds comprising a "unit farm." The cultivation systems used freshwater or saline water sources and associated algal strains, and minimal (only covering corners prone to erosion) or full, high-density polyethylene liners (the latter in the saline case only). Site-specific current and future productivity scenarios were considered. There was little certainty regarding when particular algae productivities might be achieved, so future productivities were not linked to particular years in algae scenarios as they were in the other scenarios in this report. Algae were assumed to be dewatered to a 20 weight percent (wt %) solids content.

In *BT16* volume 1, national biomass potential and minimum selling prices for the biomass were estimated for *Chlorella sorokiniana* (a freshwater strain) and *Nannochloropsis salina* (a saline strain). Current productivity-rate and future high-productivity scenarios were presented for both strains in *BT16* volume 1. In the current productivity scenarios, while the modeling was done on a site-specific basis at an hourly time step for 30 years, the mean annual biomass growth was 12.8 g/m^2/day for *Chlorella sorokiniana* and 13.8 g/m^2/day for *Nannochloropsis salina*. For the future productivity scenarios, a factor of 1.8 was used to scale up productivities on all freshwater algae cultivation sites, and a factor of 1.95 was used to scale up productivities on all saline cultivation sites, resulting in mean annual productivities of 25 g/m^2/day for both species. In *BT16* algae scenarios, biomass potentials for *Chlorella sorokiniana* in freshwater media under current productivities were estimated to be 12 million tons, 19 million tons, and 15 million tons annually for co-location scenarios with CO_2 from ethanol production plants, coal-fired electric-generating units (EGUs), and natural gas EGUs, respectively.

BT16 volume 1 included algal biomass production scenarios that used fully lined and minimally lined ponds. Ponds were lined with high-density polyethylene liners. The minimally lined ponds used liners to cover small areas at pond turns to prevent erosion. Cultivation systems that were the source of data used

Figure 12.1. | Key variables in the algae analyses in *BT16* volume 1. Full liners were not considered for the freshwater cases. The freshwater algae strain was *Chlorella sorokiniana,* and the saline algae strain was *Nannochloropsis salina*.



to develop the base case in Davis et al. (2016) had liners that covered 2% to 25% of the pond area, and the base case assumptions were used for costing capital and operating expenses in *BT16* volume 1. Only minimally lined ponds were considered for freshwater scenarios.

12.2.1 Environmental Indicators for Algae

Chapter 1 describes a set of environmental indicators that were proposed for sustainability of bioenergy systems in general (McBride et al. 2011). Most of the chapters in this report model these indicators. However, McBride et al. (2011) and a subsequent article (Efroymson et al. 2013) acknowledged that the indicators are generic and would need to be modified for particular contexts, such as algae applications.

Environmental indicators for sustainable bioenergy systems were evaluated for applicability to algal biofuels, including production processes and technologies (Efroymson and Dale 2015). Special emphasis was placed on the indicators proposed by McBride et al. (2011), which represent a focused, scientifically based, and practical set of metrics selected from a broad range of sources. Large sets of indicators recommended by the Global Bioenergy Partnership (2011) and the Roundtable on Sustainable Biomaterials (2010), as well as metrics of potential environmental impacts and resource requirements for sustainable development of algal biofuels addressed by the National Research Council (NRC) (2012), were examined. Environmental indicators for algal biomass and biofuels were selected to be practical, widely applicable, predictable in response, anticipatory of future changes, independent of scale, and responsive to management. Major differences between algae and terrestrial bioenergy feedstocks, as well as their supply chains for biofuel, were considered. Table 12.1 presents a list of 16 proposed environmental indicators for the sustainable production of algae for biofuels (Efroymson and Dale 2015); these are applicable to the estimated algal biomass potential in scenarios from *BT16* volume 1. The proposed indicators are also listed in a section on sustainability considerations for algae cultivation in the Algae Biomass Organization's *Industrial Algae Measurements* (ABO 2015).

The major categories of indicators (i.e., soil quality, water quantity and quality, GHG emissions, biodiversity, air quality, and productivity) are identical to those described in chapter 1 of this report and in McBride et al. (2011). The use of water instead of soil as the growth medium for algae means that

Table 12.1. | A Set of 16 Proposed, Generic Environmental Indicators for Modeling or Measuring the Sustainable Production of Algal Biomass and Biofuels, as Derived from Many National and International Recommendations for Sustainability Indicators, Criteria, and Standards for Bioenergy

Category	Indicator	Units
Soil quality	Bulk density	g/cm^3
Water quantity	Peak storm flow	m^3/s
	Minimum base flow	m^3/s
	Consumptive water use (incorporates base flow)	m^3/ha/day; m^3/ton; m^3/GJ (gigajoule)
Water quality	Nitrate concentration in streams (and export)	Concentration: mg/L; export: kg/ha/yr
	Total phosphorus (P) concentration in streams (and export)	Concentration: mg/L; export: kg/ha/yr
	Salinity	Practical salinity unit (PSU)

Category	Indicator	Units
Greenhouse gases	CO_2 equivalent (CO_2e) emissions (CO_2, CH_4, and N_2O)	kg CO_2e/GJ
Biodiversity	Presence of taxa of special concern	Presence
	Habitat of taxa of special concern	ha
	Abundance of released algae	Number/L
Air quality	Tropospheric ozone	Parts per billion (ppb)
	Carbon monoxide	Parts per million (ppm)
	Particulate matter less than 2.5 micrometers (μm) diameter (PM2.5)	Micrograms per m^3 (μg/m^3)
	Particulate matter less than 10μm diameter (PM10)	μg/m^3
Productivity	Primary productivity or yield	g/m^2/day or based on chlorophyll a

Modified from Efroymson and Dale (2015).

water-related indicators could be more important than soil quality indicators, such as soil organic carbon, soil nitrate, and soil phosphorus (Efroymson and Dale 2015). In contrast to the indicators proposed for terrestrial biomass, salinity is included as an environmental indicator for algal biomass production because salinity could be a concern for groundwater and surface waters if saline waters are extracted from the ground or pumped inland from the sea.

Some indicators represent a scientific consensus, whereas other indicators do not. CO_2 equivalent (CO_2e) emissions are an indicator with national and international support, and without competing proposals. Therefore, we do not discuss the advantages or disadvantages of this indicator. While consumptive water use is generally agreed to be an important water quantity indicator, many indicators and indices that incorporate regional context for water have been proposed, and some of these are discussed below and in appendix 12-A and appendix 12-B.

The context in which indicators are measured or modeled may necessitate the use of different functional units from those described above (Efroymson et al. 2013). Indicators may be expressed per biomass, per fuel gallon or gallon gasoline equivalent, per British thermal unit (Btu) impact, or per unit area, for example. Some typical functional units include fuel gallon per consumed gallon of water, fuel gallon per ton CO_2e, and consumed gallons of water per Btu. Water consumption may be expressed with respect to regional water supply or needs. Indicators are typically measured with respect to a baseline.

12.2.2 Indicators and Indices for Water Quantity—The Importance of Regional Context

The distinction between water consumption or consumptive water use (table 12.1) and water withdrawals is important to state upfront. Water withdrawn from a hydrologic system can be used for a purpose, and, depending on the use, a fraction of that water is returned into the system, where it can potentially be used for another purpose (subject to changes in water-quality attributes such as temperature and chemistry) within a short time cycle. Consumptive water use represents the water that is used and removed from the immediate hydrologic system and is not available for other uses. Consumptive water use or water consumption can be driven by evaporation or transpiration or may result from "virtual water," i.e., water that is taken up into a product, such as fruits, vegetables, beverages, etc., and transported as a commodity, often taking the water out of its basin.

Water quantity indicators go beyond the simplicity of the water consumption indicator, and even the flow indicators, described above. Regional climate, competitive uses, and valued entities (e.g., human health, rare ecological populations) are all important factors for selecting water quantity indicators. Indices are generally combinations of measured variables (indicators).

Numerous methods are available to quantify vulnerabilities in available freshwater resources at various temporal and spatial scales. Three key terms related to water-resource vulnerabilities (water scarcity, water stress, and water risk) are defined as part of the United Nations Global Compact CEO Water Mandate[1] (i.e., for chief executive officers of businesses), and with regard to bioenergy development, these must be considered holistically with all aspects of water use. Water scarcity and water stress are discussed in this chapter. Water risk, the probability and severity of an entity experiencing a deleterious water-related event, is considered a socioeconomic indicator (described as risk of catastrophe in Efroymson et al. 2016) and is outside the scope of this chapter.

- **Water scarcity:** The volumetric availability of water supply and the total use of that supply. This indicator is most often calculated as a simple ratio of total consumptive water use to the available water supply within a geographic bound, such as an individual or collection of connected basins or sub-basins. Water scarcity can theoretically be measured as often as needed and at the scales required and, accordingly, is a measure that can be compared spatially and temporally (Schulte 2014).
- **Water stress:** The ability to meet human and ecological water demand in the context of volumetric availability, water quality, environmental flows, and accessibility. Compared to water scarcity, water stress incorporates more elements beyond water supply and water use. Many methods are available for estimating water stress, and the chosen method depends on the temporal and spatial scale, the availability of data, the level of detail required, and the elements of concern for a given location (i.e., a regional study will differ from a site-specific study) (Schulte 2014).

To operationalize these indicators, the total water supply for a given geographic domain and appropriate temporal period needs to be established. In addition, current water withdrawals, consumptive use, and competing uses (including environmental flow requirements) need to be quantified at a common geographic domain and temporal period.

Several key methods are appropriate for regional- and national-scale water planning. These include the Water Resources Vulnerability Index, the Water Supply Stress Index (WaSSI) and Water Supply Stress Index Ratio, the Water Scarcity Index (Wsci), and the Water Stress Ratio.

Water resource indices are described in detail in appendix 12-A. The Water Resources Vulnerability Index, often referred to as the "withdrawal to availability ratio" (WTA ratio), is a water scarcity index and is probably the most simple and most widely used of the water resources indices. All other indices described in this chapter are variants of the basic ratio of water supply to demand (Rijsberman 2006). The WaSSI, originally proposed by Sun et al. (2008a, 2008b) and used in chapter 7, provides a measure of the relative supply and demand of water at a monthly time step for eight-digit Hydrologic Unit Code (HUC) watersheds. Despite the water-stress-related name, by the definitions herein, WaSSI is also a water scarcity index. Asheesh (2007) established the Wsci

[1] See the United Nations Global Compact CEO Water Mandate website for more: http://ceowatermandate.org/.

as a method to measure change in water availability and identify gaps that would lead to unbalanced water supply and demand in the context of a complex relationship of variables, including ecological requirements and population growth rates. This complex relationship of variables is referred to as the Water Equality Accounting System. Under the definitions laid out herein, the Wsci would be considered a holistic water stress index. Smakhtin et al. (2005) provide a simple environmental water-scarcity method, the Water Stress Indicator (WSI), which considers the relationship of water withdrawals to ecosystem water requirements.

Four key points need to be considered when using water-resource indices to evaluate environmental effects of algal biomass or bioenergy production. First, many indices use total or sector-based water withdrawal as an input, such as data that are available from the U.S. Geological Survey (USGS) 5-year water use reports (see http://water.usgs.gov/watuse/). As discussed above, water withdrawal and consumptive water use can lead to different outcomes in the volume of water available for use. Water that is consumptively used is no longer available for use in the basin or hydrologic area of interest, whereas for water withdrawals, depending on the water use sector, some portion of the withdrawn water will be returned to the system.

Second, because microalgae have growth cycles that are largely dependent on meteorological variables (primarily light and temperature), the timing of when water resources are available is critical; thus, indices that use mean annual values do not consider critical seasonal cycles, whereas indices that can incorporate a monthly evaluation are well suited to provide an appropriate level of detail. In addition, because meteorologically-induced growth cycles are also highly location-dependent, indices need to have a reasonably high level of spatial granularity to show variability, where a recommended minimum would be an eight-digit HUC boundary (see http://water.usgs.gov/GIS/huc.html).

Third, for sector-based and competitive water use assessments, indices often do not reflect required environmental flows and ecosystem requirements. Therefore, indices need to incorporate a broader-use context with respect to available supply, even if it means part of that supply remains in the river (an additional competitive water use). Assessment methods for environmental flow requirements can vary significantly in their level of detail.

Lastly, the consideration of future, altered climate, and non-stationarity effects needs to be addressed. Therefore, the use of historic long-term averages may not provide the best approach when considering potential vulnerabilities and changes to future water-resource supply. To meet future needs, the indices may need to be applied differently, but to establish a baseline, the use of historic long-term averages is appropriate.

Environmental flow is an important component of the regional context of water quantity effects of any water-use sector. Peak flow and minimum base flow are described as basic indicators for water quantity in table 12.1, but more complex measures may be needed to incorporate the regional context, some of which are specified in the water resources indices described above and in appendix 12-A. Tharme (2003) identified >200 methods available to assess environmental flows, and generally, they can be classified as hydrological, hydraulic rating, habitat simulation, and holistic methodologies. This taxonomy of environmental flow methods, as well as some of the methods themselves, are described in appendix 12-B. Indicators of environmental flow can be considered indicators of aquatic biodiversity, where flow is an important variable controlling a population or community.

12.3 Methods

In this chapter, we provide a qualitative analysis of environmental effects. Unlike most other chapters, county-level estimates of environmental indicators are not estimated.

We highlight inputs and outputs of models that were used for volume 1 and examine methods that could be used to assess particular environmental effects. For example, GHG emissions estimates are discussed for the base case pond design in the *Design and Economics for the Production of Algal Biomass* "design case study" (Davis et al. 2016), which has similar features to those in the *BT16* algal biomass scenarios and which was used for cost estimates in volume 1.[2] We also examine the modeled water consumption from the Biomass Assessment Tool (BAT) for particular scenarios and describe how those results could be put in the context of regional water use.

The BAT is the Pacific Northwest National Laboratory's integrated model, analysis, and data management suite that couples advanced spatial and numerical models to assess resource requirements, multi-criteria land suitability, site-specific biophysically-based biomass and bioenergy potential, techno-economics, and trade-off analyses (Coleman et al. 2014). With respect to production, the pond temperature and subsequent net consumptive water use (evaporation – precipitation) was modeled using a mass and energy balance model for about 88,000 potential algal production sites across the country using 30 years of hourly stochastic meteorology data and averaged across each state (Wigmosta et al. 2011). Biomass growth was modeled at an hourly time-step over a 30-year period (Wigmosta et al. 2011) for the *Chlorella sorokiniana* and *Nannochloropsis salina* parameterized with monthly temperature data.[3] Additional assumptions used in BAT are described in *BT16* volume 1, chapter 7. We describe some of the water-supply constraints here, as those are pertinent to the water quantity analysis. Annual water consumption estimates for the scenarios are presented graphically in the results section below, although monthly values were also recorded.

Sites for algae cultivation were limited, in part, by water availability (Chiu and Wu 2013; Venteris et al. 2013). In the *BT16* volume 1, a consumptive freshwater-use constraint of no more than 5% of mean annual basin flow (cumulative for sites within a watershed) helped determine the number of sites allowed (ANL, NREL, and PNNL 2012). The map that shows this initial screening of suitable sites is figure 12.2. The 5% target was based on the U.S. Environmental Protection Agency's (EPA's) established water-use rule for new thermoelectric power plants (EPA 2001). Sites were prioritized and selected based on water-use rate within the six-digit HUC until the allocated "water-use reservoir" was depleted. Because saline water resources are more plentiful, they were not constrained by required volume but rather by (1) locations where salinity ranges from 2 to 70 practical salinity units (PSU)[4], which was considered suitable

2 This design case is used as an illustration of potential assumptions. This design case would probably not be commercially scaled because it does not produce fuel that is cost-competitive with fossil fuel.

3 Strain-specific biomass productivity is a function of water temperature (minimum, optimal minimum, optimal maximum, and maximum) and light utilization efficiency of photosynthetically active radiation (PAR), both which have site-specific hourly and seasonal signals. Additional parameters that do not vary hourly or seasonally include transmission efficiency of incident solar radiation to microalgae, biomass accumulation efficiency, and others, as defined in Wigmosta et al. (2011).

4 Bartley et al. (2013) found that salinities of 22 PSU to 34 PSU provided the highest growth rates for *Nannochloropsis salina*; however, growth is possible between 8 PSU and 68 PSU. Abu-Rezq et al. (1999) found that ideal salinities for the same strain are between 20 PSU and 40 PSU. While the salinity range of 2 PSU to 70 PSU is broader than the ideal salinity target range for *Nannochloropsis salina*, it represents possible salinities that support growth of a wide range of other saline-based algae strains (Shen et al. 2015; Varshney et al. 2015; Kim, Lee, and Lee 2016). The wide salinity range also captures the uncertainties in the source data and geostatistical processing of saline water resources.

for *Nannochloropsis salina*, and (2) cultivation sites within 6.2 miles (10 km) of acceptable salinity-range groundwater or seawater sources. The constraints accounted partially for uncertainties in salinity ranges and provide economically viable water transport distances.

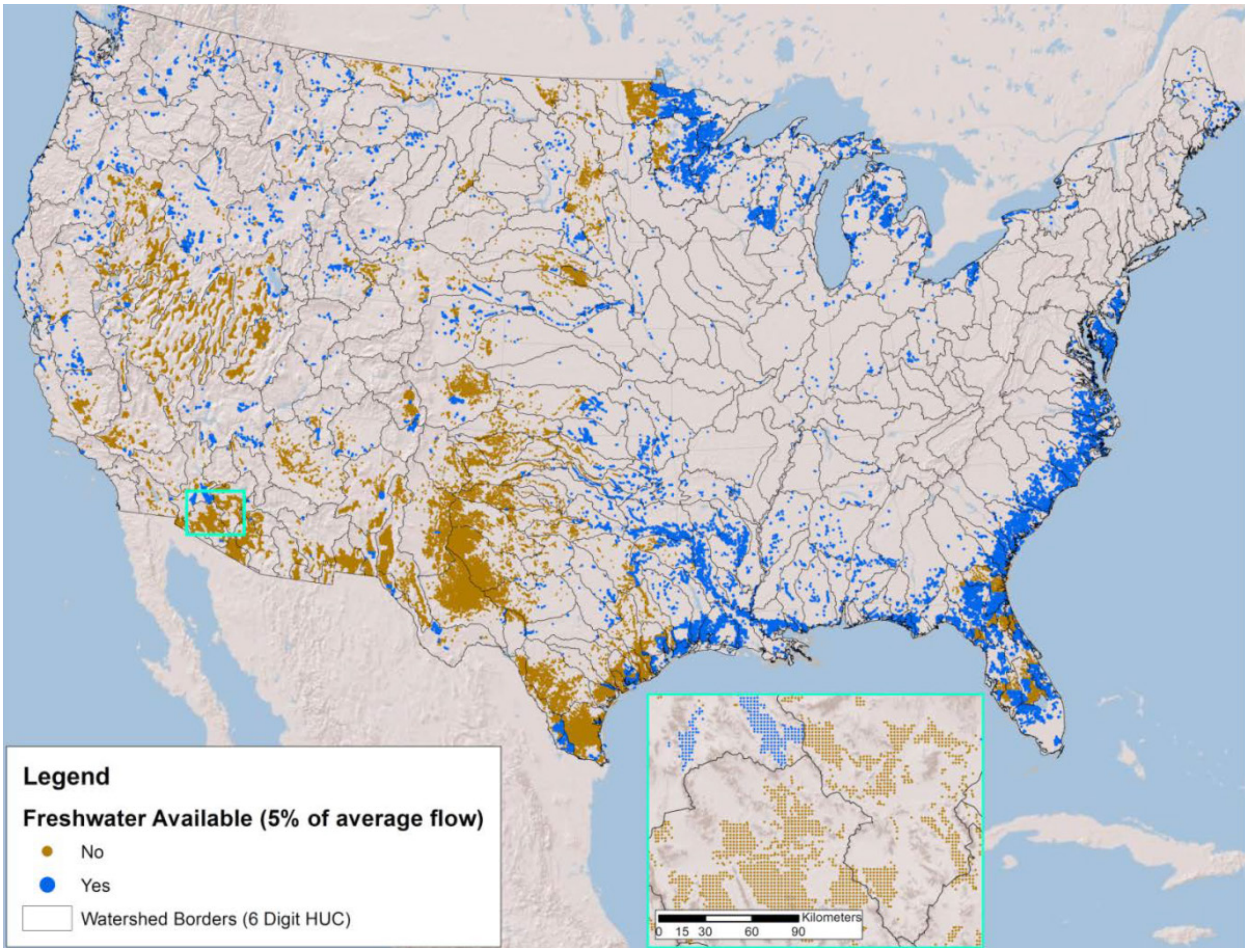
Additional siting considerations in *BT16* volume 1 related to topography and land use. For example, forest and cultivated cropland were not considered for potential algae cultivation facilities.

No simulation models were used above and beyond those applied in the resource assessment studies in *BT16* volume 1. For some indicators, we have provided estimates from relevant scientific literature and summarize variables that can affect those indicators.

12.3.1 Scope of Assessment

The variables that were considered are shown in figure 12.1 and included CO_2 co-location source, fresh or saline water, productivity, and pond liner area. All categories of environmental indicators from table 12.1 are discussed, with an emphasis on GHG emissions and water quantity indicators.

Figure 12.2. | Freshwater availability for potential algal-production sites in the conterminous United States. Water availability is determined using 5% of long-term mean annual flow at a six-digit HUC and budgeted against modeled open-pond consumptive water use at each site (Image credit: ANL, NREL, and PNNL 2012).



GHG and water quantity indicators are emphasized because (1) GHG emissions relate to many of the variables in the *BT16* volume 1 scenarios (co-location with CO_2, productivity, pond liner area), (2) water consumption was tracked in the biomass production modeling and comprised a supply constraint for algae production, and (3) most environmental analyses of algal biomass in the literature focus on GHG and water indicators. Water consumption associated with algae production in the scenarios is described quantitatively. Other environmental indicators are discussed more generally. Directional changes in environmental indicators that might result from changes in variables in the scenarios are also discussed. The methods and context of indicator measurements are described.

All scenarios from *BT16* volume 1 are considered, rather than selecting key scenarios for analysis, as for terrestrial biomass. As with other chapters, the only comparisons that can be made are among scenarios, but we cite scientific literature that makes other comparisons.

The use of photobioreactors (PBRs) is an alternative major production strategy that was not considered in the biomass estimates in *BT16* volume 1 and therefore is not addressed in detail. Pathways where algae serve as a "biocatalyst" (for example, whereby ethanol and/or hydrocarbons are secreted by cyanobacteria), were not considered in *BT16* volume 1 or in this analysis. The exclusion of these systems from the analysis does not reflect their presumed effects or lack of effects with respect to the environmental indicators described in this chapter.

12.4 Results and Discussion

This chapter is a qualitative discussion of potential environmental effects of algal biomass production with implications and uncertainties discussed in an integrated way. Formal quantitative analysis of indicators is not presented. The results and discussion sections are combined.

12.4.1 GHG Emissions

GHG emissions contribute to climate change (IPCC 2007) and related environmental and health effects, some of which are adverse (Church et al. 2013). Because the atmosphere is well mixed, effects do not depend on the counties where the gases were released or sequestered. However, climate-change effects are regional.

The primary environmental indicator of GHG emissions is CO_2e, which includes CO_2, methane [CH_4] and nitrogen dioxide [N_2O]) (table 12.1). Because the objective of this chapter is to describe cultivation-related processes that contribute to GHG emissions and other effects, we do not focus on conversion or treatment of waste products, nor do we quantify net emissions from the full supply chain. However, we discuss processes that affect GHG emissions in the full supply chain in section 12.4.1.7.

In this chapter, we focus primarily on CO_2 emissions, though we discuss other GHGs. GHG emissions from algae cultivation and dewatering are driven by the processes to capture and potentially purify and transport the CO_2, as well as additional process electricity and nitrogen and phosphorus inputs. As a general rule of thumb, to produce 1 ton of ash-free dry weight (AFDW), algal biomass requires about 1.8 tons of CO_2. The optimum CO_2 concentration for algae production depends on the strain, system design, meteorological conditions (temperature and light) and operating conditions (Zhu 2015), as well as pH.

CH_4 and N_2O are important components of CO_2e for algal biofuel systems (table 12.1) but not for the algal biomass production step of the supply chain (this analysis). The primary source of these gases is an anaerobic digestion process if used to generate power from lipid extracted algae or other process waste (Frank et al. 2012). N_2O emissions have also been measured under some algae cultivation systems, e.g., *Nannochloropsis salina* (Eustigmatophyceae) under a nitrogen headspace (Fagerstone et al. 2011), *Nannochloris* (Chlorophyta) in coastal open-pond systems emit N_2O during senescence (Florez-Leiva et al. 2010), and *Chlorella vulgaris* in high rate algal pond wastewater treatment systems supplied with nitrite in darkness (Alcántara et al. 2015). We do not quantify or discuss further potential emissions of N_2O from ponds, as these emissions would probably be small or negligible.

The consideration of the GHG impacts of algae supply chains should include the full life cycle of production through use. A number of end-use applications of algae offer the potential for substantial life-cycle GHG reductions relative to approaches that involve only cultivation. Some of these aspects of the life cycle are addressed in section 12.4.1.7.

The discussion of GHG emissions begins with the summary of a base case from the National Renewable Energy Laboratory (NREL) design case report, *Process Design and Economics for the Production of Algal Biomass* (Davis et al. 2016). Then, we discuss the benefits of co-location of algae with CO_2 sources, and in particular, the CO_2 used in the algal biomass projections in volume 1. Other variables in volume 1 scenarios (current and future productivity, freshwater versus saltwater, fully lined versus minimally lined ponds) and their potential implications for GHG emissions are also discussed. Finally, we summarize some of the important variables from the literature that have been shown to influence GHG emissions from algae cultivation or dewatering systems. These include variables that affect energy return on investment (EROI).

12.4.1.1 An Algae Base Case

Analogous to the algae culture-system design and costs in *BT16* volume 1, which modify costs in Davis et al. (2016), the GHG emissions discussion in this chapter begins with the GHG emissions estimate from *Process Design and Economics for the Production of Algal Biomass* (Davis et al. 2016). Departures from Davis et al.'s (2016) assumptions in *BT16* volume 1 regarding unit farm size, pond size, species, and resource assumptions are discussed in detail in volume 1 and summarized below.

Davis et al. (2016) assume a freshwater open-pond/raceway cultivation system with an assumed cost that represents the average cost of four 10-acre pond designs, and, unlike the strains assumed in this analysis—*Chlorella sorokiniana* and *Nannochloropsis salina*—they project productivities and GHG emissions for *Scenedesmus acutus (LRB-AP 0401)*. Davis et al.'s (2016) base case and most of the algae scenarios from *BT16* volume 1 assume a minimal liner that covers 2%–25% of total pond area in the four pond designs from which Davis et al.'s (2016) base case is derived. Davis et al. (2016) assume in-ground gravity settlers, followed by hollow fiber membranes and centrifugation to concentrate or dewater the harvested biomass; however, they note that the dewatering performance represents research and development advancement goals to meet cost targets.

We assume the same inoculum technology, water-circulation pipelines, average pond-circulation power demand, and product storage tanks as in Davis et al. (2016), and therefore, the same GHG emissions for these components. As in Davis et al. (2016), biomass is harvested and processed through three dewatering steps—gravity settling, hollow fiber membranes, and centrifugation—to concentrate the biomass from 0.5 g/L (0.05 wt % AFDW) to 200 g/L (20 wt %) in the product stream.

Davis et al. (2016) assume that purified CO_2 from flue gas carbon capture is captured (amine scrubbing, membrane purification, etc.) and delivered to the unit farm (cultivation system) via pipeline under high pressure, is stored in pressurized spherical storage tanks, and is distributed and sparged into individual ponds during daytime production. The *BT16* scenarios assume that cultivation is co-located with existing natural gas power plants, coal-fired power plants, or ethanol plants, and CO_2 is sourced via a low pressure, direct flue gas feed. While CO_2 concentrations vary depending on the source and thus will impact transport efficiencies, the movement of non-stripped flue gas can reduce capital equipment needs and lower the parasitic power load (energy used for internal purposes rather than exporting) from the power plants. For non–power plant CO_2, a reduction in imported energy could be expected.

As Davis et al. (2016) note, "Both the CO_2 input and 'emissions to the air' from the cultivation ponds (attributed to CO_2 retention efficiency losses) are treated as biogenic in nature, following accepted methodologies for CO_2 accounting in algal biofuels LCA [life-cycle-analyses] which dictate that although the CO_2 originates from fossil power plant flue gas, the power plant is operated to generate power and not to provide CO_2, which otherwise would be emitted to the atmosphere and then later could be utilized in dilute form as biogenic CO_2 for growing a different biomass resource" (Frank et al. 2011). Biogenic CO_2 does not add to GHG in the atmosphere (Karlsson and Byström 2010) and is not accounted for in the Intergovernmental Panel on Climate Change global warming methodology (Fisher et al. 2007). CO_2 sourced from an ethanol production plant would also be biogenic, since biomass is processed to produce it.

The GHG emissions for the base case (ending with the partial dewatering, as described above) in Davis et al. (2016) are estimated at 0.73 ton CO_2e/ton AFDW biomass, with 0.38 ton CO_2e/ton AFDW biomass representing emissions due to carbon capture from flue gas and 0.30 ton CO_2e/ton AFDW biomass representing process electricity. Given that the process of CO_2 transport assumed in *BT16* should not use as much compression energy (or related electricity) as monoethanolamine carbon capture, the associated CO_2e/ton algal biomass in the *BT16* system should be lower than emissions in Davis et al. (2016). One algal biofuel company that is located adjacent to a coal-fired power plant from which it obtains CO_2, captures, and delivers CO_2 without compression, with GHG emissions reported on the order of 0.03 ton CO_2e/ton AFDW algal biomass.

The life-cycle inventory on which GHG emissions in Davis et al. (2016) are based includes values for biomass; nutrient, water and electricity demands; water and biomass lost to blowdown (i.e., pumping water exchange); and water, CO_2, and O_2 emissions, with the energy to capture CO_2 estimated separately (Davis et al. 2016, table 20). The quantity of GHG emissions is driven primarily by the processes to capture, purify, and transport the CO_2, and also includes process electricity (U.S. average electricity mix from the grid, 0.65 kg CO_2e/kWh) and ammonia and diammonium phosphate nutrient inputs. The inputs and outputs are not presented here to avoid confusion with the site-specific analysis that was performed in *BT16* volume 1 and resulted in biomass, water, and CO_2 used that were driven by meteorological variables. So, while Davis et al. (2016) provide a good starting point to estimate GHG emissions, they do not provide regionally specific GHG emissions.

Some differences between the assumptions in this chapter (taken from *BT16* volume 1) and in Davis et al. (2016) affect GHG emissions from algal biomass production and logistics processes for the current or future productivity cases. Scenario differences in *BT16* are summarized in table 7.5 of volume 1. *BT16* scenarios include 100 10-acre ponds per facility, rather than Davis et al.'s 500 10-acre ponds; *Chlorella sorokiniana* and *Nannochloropsis salina*, rather than mid-harvest, high-carbohydrate *Scenedesmus acutus*; site-specific current and future productivities, rather than a cultivation productivity target; and saline media for some of the scenarios instead of just freshwater.

Some of the differences between scenarios in *BT16* and Davis et al. (2016)—for example, productivity estimates—relate to the different purpose of *BT16*, which is to estimate current and future national biomass potential, compared with that of the cultivation design case report, which is to describe "aspirational" targets for future facilities. For the current productivity scenarios in *BT16* volume 1, we assumed lower site productivities than the target in Davis et al. (2016).

The use of saline water affects estimates of GHG emissions. We consider scenarios that assume that ponds are fully lined if saline water is used. However, we recognize liners are not a requirement for every location, so we also consider scenarios wherein saline ponds are minimally lined, as with freshwater, with the objective of controlling pond erosion. The use of injection wells is assumed for media disposal under all saline water scenarios. Note, however, that other saline-disposal options exist, including evaporation ponds and landfill-style disposal of salt, discharge to a water-treatment facility, and, in the case of coastal sites, cleanup and discharge to the ocean (Mickley 2001). Additionally, there are beneficial uses for saline concentrate including oil-well field injection, solar ponds, aquaculture, wetland creation/restoration, and high-value salt and chemical products (GEO-Processors USA 2006; Jordahl 2006).

In addition, the *BT16* scenarios do not reflect the carbon capture and compression assumptions from Davis et al. (2016). Instead, the CO_2 in *BT16* is transported from sources to the algae production sites (unit farms) using pipelines and blowers, which have a smaller (but unquantified) energy footprint and GHG emissions. The specifics of co-location are described below.

12.4.1.2 Co-Location with CO_2

Table 12.2 shows the CO_2-related benefits for power plants from the scenarios in *BT16* volume 1, including the total CO_2 used in the algae production scenarios and the percentage of the total across the conterminous United States. These quantities of CO_2 are largely utilized by algae with a fraction released to the atmosphere through pond outgassing. This fraction is assumed to be 18% in *BT16* volume 2 and 10% in Davis et al. (2016). (Atmospheric release of CO_2 for PBR systems would be minimal).

Co-location of algae with CO_2 in these scenarios should delay CO_2 emissions to the air and provide a beneficial use of the CO_2 compared to scenarios in which the gas is emitted directly to air. In addition, the potential displacement of fossil-based fuels with algae-sourced fuels should be considered in the estimate of overall CO_2 reduction. Although CO_2 can be temporarily used by algae (Menetrez 2012), the decomposition rate of waste biomass and recycling is an important consideration for determining the quantity and rate of emissions (Fernandez et al. 2012) (See section 12.4.1.7). Moreover, waste CO_2 utilized by algae in the *BT16* scenarios might be released to the atmosphere more rapidly than if the CO_2 were captured and sequestered in an underground geological formation (carbon capture and sequestration [CCS] technology). The additional power (i.e., parasitic power load) required for CCS is substantial (20%–30%); thus, more power needs to be generated to maintain contracted electricity exports. The CCS approach then requires more energy resources, such as coal or natural gas, and ultimately generates more CO_2. This scenario needs to be considered in the context of the reference case.

The energy and infrastructure required to capture and transport impure CO_2, as in *BT16* volume 1, can be substantial. However, large capital and energy costs and related GHG emissions associated with purifying and compressing the CO_2 are avoided.

The GHG emissions estimate from Davis et al. (2016) above was presented with the caveat that an important factor determining the GHG emissions is the CO_2 carbon-capture technology occurring off-site, which is based on current technology, i.e., monoethanolamine carbon capture. This accounted for 55% of the emissions. This emissions output should improve with co-location with CO_2 sources considered in volume 1 or second-generation carbon-capture technology.

Low compression of CO_2 with blowers, as well as

Table 12.2. | Summary Results for Potential Algal Biomass from CO_2 Co-Location with Ethanol Production Plants, Coal Power Plants, and Natural Gas Power Plants Using *Chlorella sorokiniana* (Freshwater) or *Nannochloropsis salina* (Saline) under Current and Anticipated Future Productivities

	Chlorella sorokiniana			*Nannochloropsis salina*		
	Ethanol production	Coal EGU	Natural gas EGU	Ethanol production	Coal EGU	Natural gas EGU
Current productivity						
Total annual biomass (million tons/year)	12	19	15	10	54	21
Total cultivation area (thousand acres)	905	1,257	790	793	3,349	1,096
Total CO_2 used (million tons/year)	29	46	37	25	134	52
Percentage of total CO_2 in conterminous United States used in co-located algae production	19.3%	1.7%	8.9%	16.8%	4.91%	12.6%
Average distance from CO_2 source to algae facility (miles)	15.2	6.2	4.8	16.0	8.9	6.7
Future productivity						
Total annual biomass (million tons/year)	13	10	--	11	12	--
Total cultivation area (thousand acres)	508	257	--	435	299	--
Total CO_2 used (million tons/year)	32	25	--	28	30	--
Percentage of total CO_2 in conterminous United States used in co-located algae production	21.3%	0.9%	--	18.5%	1.1%	--
Average distance from CO_2 source to algae facility (miles)	14.5	3.8	--	14.6	4.4	--

These analyses are limited to the conterminous United States.

EGU is electric generating unit

piping, varies with productivities, strains of algae, and the co-location scenarios, i.e., the concentration of CO_2 in flue gas and distances from source to facility. Both compression and piping will influence CO_2 emissions. CO_2 outgassing losses are usually higher for sparging low-pressure, nitrogen-containing flue gas into pond sumps (15%–25% losses), than sparging high-pressure, purified gas into sumps (10% losses), according to Bao et al. 2012, de Godos et al. 2014, and Davis et al. 2016. This variability in CO_2-use efficiency was not captured in *BT16* volume 1, which assumed 82% utilization efficiency that was directly related to variability in biomass growth and subsequent CO_2 demand.

12.4.1.3 Productivity

Increases in productivity are associated with improved efficiencies and more energetically favorable cultivation systems than lower productivities (Sills et al. 2013). Energy requirements associated with plastic liner manufacture, mixing, sparging CO_2, water transport, and dewatering are lower per unit biomass as areal productivity increases (Sills et al. 2013). Moving the water to and from the dewatering step is energy and CO_2 intensive (Frank et al. 2011; Weschler et al. 2014), so moving less water per unit biomass is advantageous with respect to GHG emissions on a biomass or biofuel basis.

12.4.1.4 Saline versus Freshwater

We do not know of a study that has modeled or measured GHG emissions from saline and freshwater culture media under similar conditions. While freshwater is easier and less expensive to access, it has a higher rate of evaporation and, thus, a greater consumptive use of water. There is an opportunity for water recycling, which will ultimately help minimize pumping energetics, as opposed to continually drawing the full, required water volume from a clean source. Algae production using brackish, saline, or hypersaline waters (dependent upon the requirements of specific algal strains) requires water sources that can be more energy intensive to access (e.g., deep saline groundwater). Near-coastal sites may be an exception. These ponds must maintain specific salinity ranges for optimal biomass production, requiring new water to be pumped in and old water expelled (i.e., blowdown). In general, required blowdown water and associated pumping energetics will decrease with an increase in the strain-specific allowable pond salinity concentration.

For both freshwater and saline-based ponds, there is a need to dispose of the water. Freshwater can be recycled back to the production ponds where remnant nutrients can be made available for use, or water can be treated and discharged according permitted regulation (e.g., National Pollutant Discharge Elimination System permit). Saline water will require one of several options for disposal, as noted previously, some of which could be more energy- and cost-intensive (e.g., re-injection wells, ocean outfall) than freshwater disposal, and some less energy-intensive but not necessarily less cost-intensive (e.g., evaporation ponds).

12.4.1.5 Full Plastic Liner versus Minimal Plastic Liner

Energy is required to produce the plastic for a polyethylene pond liner. This energy translates to GHG emissions. While some energy could be required to make unlined ponds suitable for cultivation (e.g., compaction, movement of clay, or addition of carbon source to promote microbial clogging), this energy, and the associated GHG emissions, should be lower than that associated with lined ponds.

Canter et al. (2014) studied infrastructure-associated emissions for renewable diesel production from algae in ponds based on a design by Lundquist et al. (2010) and found that unlined ponds showed a 39% decrease in GHG emissions compared to the baseline high-density polyethylene (HDPE) pond liner design. Even if plastic liners were only used to cover berms to protect against erosion, this infrastructure accounted for a large fraction of infrastructure-related pond emissions. GHG emissions were highly sensitive to pond liner thickness and material lifetime. As Canter et al. (2014) note, “the first step to reducing infrastructure-cycle emissions would be to reduce or eliminate pond liners if soil conditions and environmental regulations permit.” The material lifetime would be an important determinant of emissions.

Liners also affect hydrodynamics and the energy required for mixing. The hydrodynamics are related to roughness coefficients (Chow 1959). Energy differences translate into differences in GHG emissions.

Moreover, excavating and moving soil or covering soil with a liner would be expected to affect the carbon dynamics of soil and associated GHG emissions, compared to unlined ponds or relatively undisturbed soils. However, these potential effects have not been studied.

12.4.1.6 Highlights of the Literature on GHG Emissions, Energy Inputs, and Land-Use Change Related to Algae Cultivation

In this subsection, we highlight some of the literature on factors that influence GHG emissions from cultivation in ponds, the resources and infrastructure needed for cultivation, and dewatering processes. These are factors that are not considered in the comparisons above. Literature on net energy inputs (i.e., energy return on investment [EROI]) is also considered, because energy use—particularly fossil energy use—affects GHG emissions.

Losses of CO_2 from open ponds influence net emissions. Both Davis et al.'s (2016) and the *BT16* volume 2 estimates of CO_2-use efficiency and outgassing would change with productivity, pH, temperature, and water flow changes associated with the *BT16* scenarios, in addition to changing with the CO_2 concentration in flue gas, as discussed above. White and Ryan (2015) note that Sapphire Energy's placement of CO_2 diffusers within a sump for high CO_2-use efficiency would not be feasible at a scale of thousands of acres, because of the tendency of sumps to fill with sand and silt.

Factors affecting EROI of algal biofuel production were described in an evaluation of socioeconomic indicators of algal biofuels (Efroymson et al. 2016). The subset of factors that apply to cultivation of algae in ponds and subsequent dewatering are presented in table 12.3. A theme in the literature is that environmental and economic costs of biofuel production are more favorable when microalgae or cyanobacteria are produced using renewable energy sources, such as solar, wind, or on-site biomass-generated methane for electricity production (Beal et al. 2015). Passell et al. (2013) found that increasing productivity and scale decreased the net energy ratio (energy in/energy out).

GHG-emissions indicators also reflect land-use change (LUC) that would be attributable to algal biofuel systems. Land converted to algal biomass production systems is expected to include industrial brownfields, rangelands, deserts, abandoned or unproductive farmland, dredge spoil islands, or other coastal areas (NRC 2012). The production system could decrease soil carbon sequestration to an extent that would depend on the CO_2 storage associated with the baseline land condition and the surface soil that was excavated. Arita et al. (2016) found that including the contribution of the direct LUC associated with carbon stocks disturbed by algae facilities would mean that some of the suitable siting locations for algae facilities from the scientific literature (based on GHG emissions criteria) would no longer meet the net emissions benefits criterion for advanced biofuels under the U.S. Renewable Fuel Standard (RFS2).

Indirect LUC could result from algae production if land management is altered on distant land as a result of algae production, and that could have GHG implications. Variations in the definition of indirect LUC are addressed in chapter 3. Cropland is not used to cultivate algae in the *BT16* volume 1 scenarios, so the probability that algae production would lead to the transition of forest or other land to cropland is low. Pastureland is not excluded from the transitions in *BT16* volume 1, however. If protein coproducts were produced, algae production could allay potential concerns about food-related LUC, because land area required to produce protein could be reduced.

12.4.1.7 GHG Emissions and the Supply Chain

The discussion above, as well as the scenarios in *BT16* volume 1, pertains to biomass potential from cultivation and dewatering. Including the whole sup-

Table 12.3. | Factors Affecting Energy Return on Investment for Production and Dewatering of Algal Biomass

Infrastructure	Installation of ponds
	Geometry of pond/raceway (e.g., baffles)
	Pond liner[1]
	Mixing method (e.g., paddlewheel assumptions, airlift pond circulation)
	Number, type, and size of pumps or gravity-fed volume transfers
Resource Requirements	Fertilizer (embodied energy, recycling)
	Source and purity of CO_2 and distance to source (e.g., flue gas)[1]
	Technology for purifying CO_2
	Wastewater use
	Rate of sparging of CO_2
	Source of water and delivery (drilled wells or pipeline)
Cultivation	Areal growth rate, including improvement by species selection, genetic modification or enhanced growth conditions[1]
	Algal strain—lipid composition and properties, such as ability to settle[1]
	Temperature control system
	Use of artificial lighting at night (or not)
	Storage of flue gas (or not)
	Recirculation of water
Processing	Pre-harvesting with settling ponds
	Harvesting (e.g., filtration, flocculation, flocculant choice, centrifugation)
	Dewatering, drying (including source of heat)
Other Energy Credits	Quantity and type of coproducts, if included in system boundary
	Wastewater treatment credits (and aeration energy offsets)

Table is modified from table 4 in Efroymson et al. (2016), which includes the references.
[1] Variable in *BT16* algal biomass scenarios

ply chain would allow more complete consideration of environmental indicators such as CO_2e throughout the life cycle. Compared to terrestrial biomass, algae cultivation is more tightly integrated with downstream fuel production processes. The purpose of this chapter is not to review life-cycle analyses; however, some early GHG life-cycle analyses for algal biofuels were reviewed by the NRC (2012).

To include downstream processes involves making assumptions about later steps in the supply chain, such as (1) assuming a conversion process, fate of waste products, or target fuel quantity, as in the U.S. Department of Energy's (DOE's) "design cases," or (2) assuming particular technical and economic criteria for "state of technology" analyses.

For example, Argonne National Laboratory conducted supply-chain life-cycle analyses for a hydrothermal liquefaction and upgrading conversion pathway to estimate seasonal energy use and GHG emissions associated with renewable diesel production (Pegallapati et al. 2015). The material and energy intensity of the biomass-conversion step was taken from Jones et al. (2014). GHG emissions were from fuel combustion, fertilizer production (used for cultivation), energy for pumping biomass from the harvesting process to the biorefinery, and other processes. Conversion was assumed to be co-located with cultivation, which is the norm for algae, so transportation fuel to the conversion facility was not needed. These emissions are sensitive to how emissions are allocated to coproducts in life-cycle analyses (Wang et al. 2011). The model used was a version of the Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation (GREET) model used for GHG analysis of terrestrial feedstocks in chapter 4 of this volume. The conversion facility was sized based on an average algae productivity of 30 g/m^2/day, which is close to that of the future-productivity scenarios in *BT16* volume 1. This study concluded that algae renewable diesel has lower GHG emissions, fossil energy use, and petroleum use than does petroleum diesel under the conditions in that report, which included the use of flue gas, rather than captured CO_2 (Pegallapati et al. 2015). Further analyses by Argonne National Laboratory (Frank et al. 2016) for lipid extraction and thermal processing pathways used in three scenarios showed that GHG emissions satisfied the RFS2 for advanced biofuels only when flue gas transported under low pressure was the source of CO_2, as in *BT16*.

In the Davis et al. (2016) base case for algae cultivation in ponds that was described earlier in this section, the authors describe recycle and consequent GHG benefits that could accrue if the downstream conversion process were integrated with the cultivation process. These include the following:

- "Recycle of a fraction of CO_2 anywhere between 10%–40% (depending on downstream conversion steps and yields), which would reduce incoming makeup CO_2 demands by a similar fraction."
- "Recycle of a substantial fraction of nutrients on the order of 50% P and 50%–90% N (dependent on similar factors as noted for CO_2)."
- "In some cases, the generation of a net electricity coproduct in the downstream conversion facility would partially offset the power demand . . . for the biomass production facility."

The fate of the algal biomass is important when estimating GHG emissions. Remaining biomass after oil extraction (in the lipid-extraction pathway) can be used for products such as fertilizer or animal feed. Biomass can be fermented to produce ethanol or pyrolyzed to create oil, gas, and char. Whether the whole biomass is directly converted to biocrude via hydrothermal liquefaction (Elliott et al. 2015) or lipid-extracted biomass is anaerobically digested to produce CH_4 for electricity production, the resulting digestate can be recycled to provide microalgae-required nutrients or applied as a soil treatment. Frank et al. (2012) found that GHG life-cycle analyses were highly dependent on biogas-production parameters, including "yields from digesters, yields from gasification, fugitive emissions, nutrient recovery rates, and electrical efficiency of the [combined heat and power] generator." CH_4 and N_2O emissions contributed substantially to total GHG emissions when the anaerobic digestate was applied to soil. Luo et al. (2010) assumed that annual disposal of cyanobacteria biomass would be via deep well injection, which could result in a slight net GHG reduction for the PBR system.

Two algal biofuel pathways that involve the use of cyanobacteria as catalysts that secrete ethanol in PBRs were approved by EPA for generation of advanced biofuel Renewable Identification Numbers under RFS2. These pathways are not pertinent to the open pond/raceway cultivation systems that were the focus of *BT16* volume 1, but they are relevant to this discussion of supply chain strategies to reduce GHG emissions. The pathways include the Helioculture Sunflow-E ethanol process of Joule Unlimited Technologies, Inc., which is estimated to reduce life-cycle GHG emissions compared to the statutory petroleum baseline by 85% (Grundler 2016), and the

Algenol Direct-to-Ethanol Process, which is estimated to reduce life-cycle GHG emissions compared to the statutory petroleum baseline by 69% (Grundler 2014).

12.4.2 Water Quantity

As is noted in chapters 7 and 8, freshwater availability is declining in some regions as a result of increased water demand for irrigation, power generation, and domestic water use, in part because of a growing population, and partly as a result of altered climate patterns. Tracking water resource use for algal biomass cultivation is the first step toward determining effects on water availability and water-related effects. This section discusses water consumption, as well as indicators and methods that place that consumption in the context of competitive uses of water and the regional environment.

Most commonly, microalgae feedstock production occurs in open raceway ponds that can be operated with either fresh or saline water sources, depending on the algal strain. These are relatively shallow ponds (30 cm deep in the scenarios in this chapter) with a large surface-to-depth ratio designed to maximize capture of sunlight and minimize “dark zones” that may result in loss of biomass through dark respiration. Cooling of the open ponds is generally achieved through evaporation (a significant source of consumptive water use), and warming occurs through solar-radiation inputs. The thermal mass of water and surrounding soil provide some buffering against rapid changes in pond temperature; thus, pond temperature fluxes are not as rapid as changes in air temperature. A complex relationship exists between the pond water temperature, hourly meteorological data, optimal operating conditions for maximal biomass productivity, and evaporative loss of water.

Environmental indicators include water quantity indicators and water quality indicators. Water quantity indicators are emphasized because (1) the scenarios in *BT16* volume 1 tracked water quantity, and (2) more research has focused on effects of algae production on water quantity than on potential impacts to water quality.

Consumptive water use from pond systems is affected by algae cultivation operations and varies geographically. Cultivation systems with summertime high temperatures and low humidity have higher rates of evaporation, greater pond cooling, and consequently, higher rates of water consumption. Alternatively, cultivation systems located in regions with high summertime temperatures and high humidity have lower evaporation rates, less water use, and limited capacity to cool. The selection of algal strains that can operate under site- or region-specific, seasonal environmental conditions can provide for more favorable ratios of biomass production to water consumed.

Consumptive water use is a system-specific indicator that alone does not capture local availability and competing uses (NRC 2012). Peak storm flow and minimum base flow (table 12.1) are ecosystem-related indicators of water quantity. Competing uses are discussed below, as well as more complex methods for assessing water-quantity-related effects that integrate local availability and environmental water requirements with system withdrawals and use.

This section on water quantity describes water consumption that was estimated in the biomass production modeling in *BT16* volume 1. Water consumption estimates are placed in the context of the variables in the scenarios, namely, productivity, freshwater vs. saltwater, and full liner vs. minimal liner. The source of co-located CO_2 is not thought to influence water withdrawals or consumption. Water consumption is discussed in the context of competitive use and regional availability.

Water is a regional resource. Therefore, the discussion of the environmental effects of algal biomass production with respect to water includes a discussion of the national context of water use and water availability. This emphasis contrasts with that of GHG emissions. While the regional environment can affect net GHG emissions (e.g., via land management–related changes in soil organic carbon and temperature effects on CO_2 use), the implications of GHG emissions from a region are global. The effects of algae cultivation systems with respect to existing water sources, competing use, future demand, and water quality are regionally variable.

12.4.2.1 The Context of Water Use

The increased interdependencies of energy, socioeconomic variables, and environment related to available water resources are magnified by higher variability in inter-annual climate, extreme events, non-stationarity, and spatiotemporal migrations of climate (Skaggs et al. 2012). Within the energy-production domain alone, demand for water resources is rapidly growing, as various types of energy production (thermoelectric, hydroelectric, hydraulic fracturing for natural gas, bioenergy, coal, etc.) continue to increase (DOE 2006; Bauer et al. 2014; McMahon and Price 2011). Water use for algae production needs to be evaluated in this larger context of the food-energy-water nexus, particularly since freshwater withdrawals for agriculture represent about 32% of all freshwater withdrawals, and thermoelectric power generation represents about 45%, totaling about 78% for these two sectors (Maupin et al. 2014). Between conveyance and consumptive use by the crops, irrigated agriculture (fresh surface water and groundwater) has the highest rates of national water use, where ~80% of water withdrawn is consumed, and 20% is provided as return flows (Solley et al. 1998). Wigmosta et al. (2011) review the water intensity of transportation fuels in volume of water per distance driven, based largely on King and Webber (2008). Biologically based transportation fuels typically consume much more water than petroleum-based gasoline.

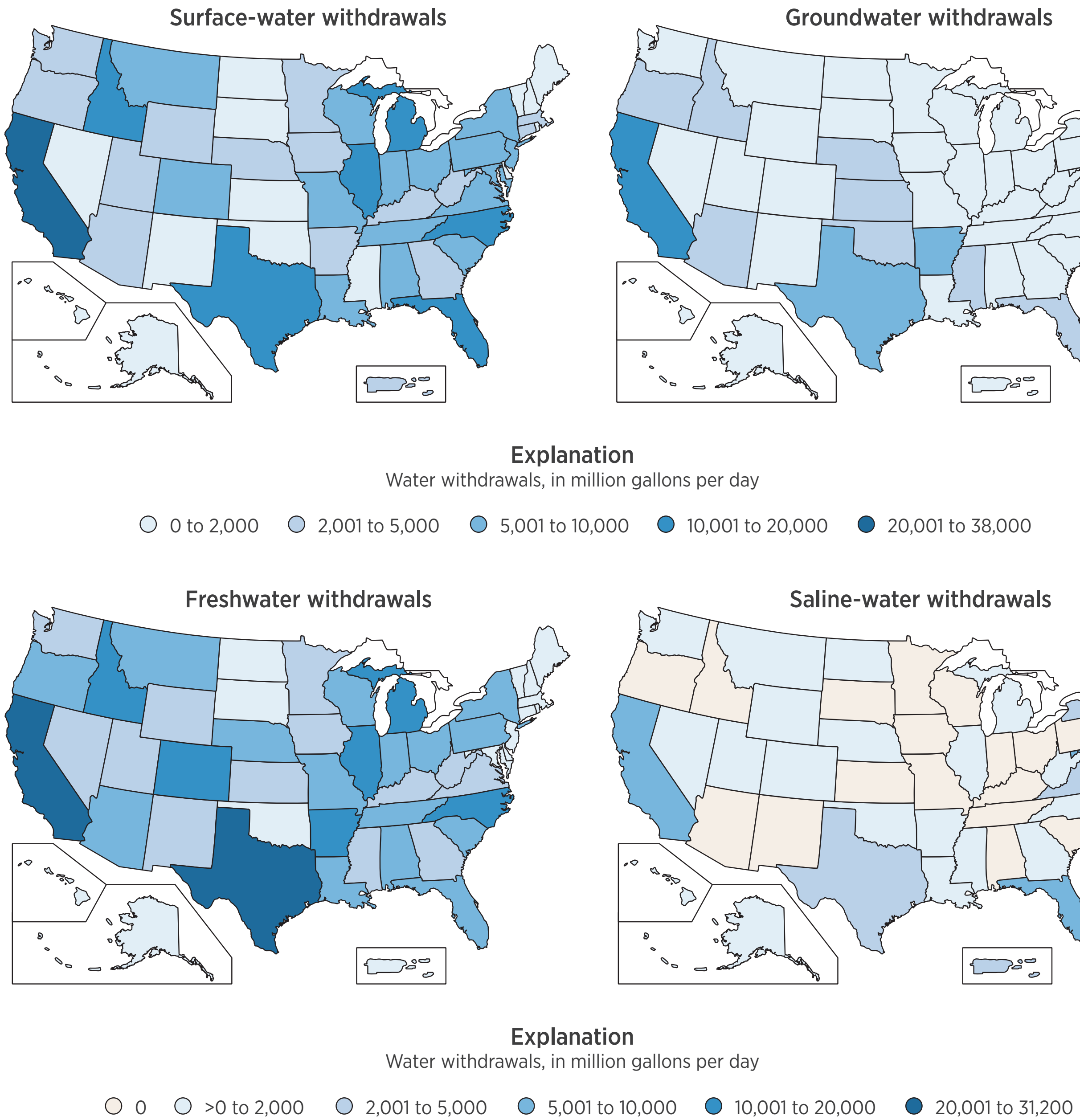
In the context of water-resource competition, Moore et al. (2015) note that "…available water resources [are] understood by evaluating the quantity, timing, and spatial distribution of water availability and use. The location and timing at which water is available and consumed dominantly affects the extent to which not only energy and water influence one another, but also the greater cross-sector dependencies that, for example, influence agriculture, industry, environment, economics, and social well-being."

To help describe the water resource landscape, the USGS produces a county-scale U.S. water-use report at 5-year intervals and provides sector-specific water-use information, including irrigation/agriculture, domestic, industrial, thermoelectric, livestock, and mining sectors (Kenny 2009; Solley et al. 1998; Maupin et al. 2014). It is important to note that after 1995, these reports only include water withdrawals and not consumptive water use. The distinction is critical, as described above. To provide context on surface and groundwater use for both fresh and saline water, Maupin et al. (2014) provide a state-level look of average daily withdrawals from 2005–2010 (fig. 12.3).

Maupin et al. (2014) estimated that for average water withdrawals over 2005–2010 (surface and groundwater), 86% were freshwater sources, and the remaining 14% were saline. Of the total withdrawals, 78% of the water resources (freshwater or saline) came from surface water resources (84% freshwater, 16% saline). Groundwater sources represented 22% of total withdrawals with 96% being freshwater and 4% saline. States with the nation's majority of withdrawals include California, Texas, Idaho, Florida, Illinois, North Carolina, Arkansas, Colorado, Michigan, New York, Alabama, and Ohio.

Appendix 12-C shows the fractional contributions of sectors to total consumptive water use. Most competitive uses are for freshwater; competitive use of the saline water supply is primarily related to thermoelectric power plants.

Figure 12.3. | The USGS provides a state-level look at withdrawals of fresh and saline water for both surface and groundwater sources averaged during 2005–2010 (Image credit: Maupin et al. 2014).



12.4.2.2 Water Consumption in Scenarios from BT16 Volume 1

Saline versus Freshwater—Qualitative Discussion

Both saline and freshwater scenarios were included in *BT16* volume 1. Differences in the use and operations of freshwater and saline water result in tradeoffs between the two types of sources. Saline and freshwater sources differ with respect to availability, access and transport cost, competitive use, maintenance, and disposal needs and costs.

Sources of freshwater are generally easier and less expensive to access than saline water sources. Saline resources are not as well characterized as freshwater resources. The last nationwide saline water assessment was conducted by the U.S. Geological Survey (USGS) in 1965 (Feth et al. 1965). More recent characterization of these water sources has been supported by carbon-sequestration efforts, though the emphasis has been on very deep saline groundwater reservoirs (>800 m depth) that would not be economically viable as a source for algae production (Venteris et al. 2013).

Freshwater has high competitive use and is in limited supply in many parts of the country; however, saline water sources (brackish, saline, and hypersaline) from groundwater or seawater are abundant resources with lower competitive use than freshwater but typically require more energy to transport from source to pond. In addition, the ion chemistry in saline groundwater is highly variable, and sources need to be screened for toxicants and composition compatible with specific algal strains (Venteris et al. 2013). For example, a total dissolved solids (i.e., salinity) characterization of existing, produced water (i.e. oil and gas) wells in the conterminous U.S. is shown in figure 12.4, where variability in salinity ranges from 1 to 400 practical salinity units (PSU) (1,000–400,000 mg/L). Seawater ion chemistry is more consistent with salinity between 33 and 37 g/kg.

For freshwater open-pond cultivation systems, the water systems must be maintained to compensate for net losses (evaporation minus precipitation); whereas, for saline open-pond systems, water is maintained not only for the volume of water, but more importantly, to maintain the salinity required by the cultivated algal strain (see fig. 12.5). Freshwater open ponds have a higher rate of evaporation than do saline sources; however, freshwater can be treated and recycled for further use, reducing pumping costs associated with bringing clean water to the site. In order to maintain salinity targets, saline pond systems can require 2-3 times the amount of withdrawn water that freshwater ponds require, depending upon the salinity target, source water salinity, and local meteorological conditions (Venteris et al. 2013). Note that this water use is not required to replace evaporative loss, but rather, for pond blowdown, where a fraction of the pond water is discharged and replaced with new water to keep an ideal pond operating environment for the strain of interest.

Saline water concentrate must be disposed after blowdown. The most commonly considered option for saline groundwater is through re-injection wells and for seawater is a marine outfall that may or may not be the same pipe construction that draws in source water. Other alternative saline-disposal options exist, including evaporation ponds and landfill-style disposal of residual salt, discharge to a water treatment facility, oil well-field injection for secondary oil and gas recovery, solar ponds, aquaculture, wetland creation or restoration, and high-value salt and chemical products (GEO-Processors USA 2006; Jordahl 2006; Mickley 2001).

In general, the use of saline water resources comes with an increased capital and operational expense, as compared to freshwater. An example is the $32 minimum selling price per ton of biomass added for blowdown waste disposal for saline systems in *BT16* volume 1 and Davis et al. (2016). Other increased capital and operating expenses for the use of saline resources (e.g., deeper wells to access water, corrosion-tolerant construction materials) were not considered in *BT16* volume 1. From the perspective of production facility siting, managing for salinity concentrations can increase operational expenses.

From the perspective of operations and operational expenses for saline water sources, the relationship of salinity concentration to water use (evaporative loss and blowdown) affects the quantity of water that needs to be moved to the site (fig. 12.5). Higher pond target salinities require less blowdown and lower inputs of new saline water. The salinity of the water source is also important; a low salinity water source requires less blowdown and lower inputs of new saline water to maintain pond target concentrations.

Figure 12.4. | Oil and gas wells with associated total dissolved solids measurements from produced water. These well locations do not include recent hydraulic fracturing wells. (Data source: Breit 2002).

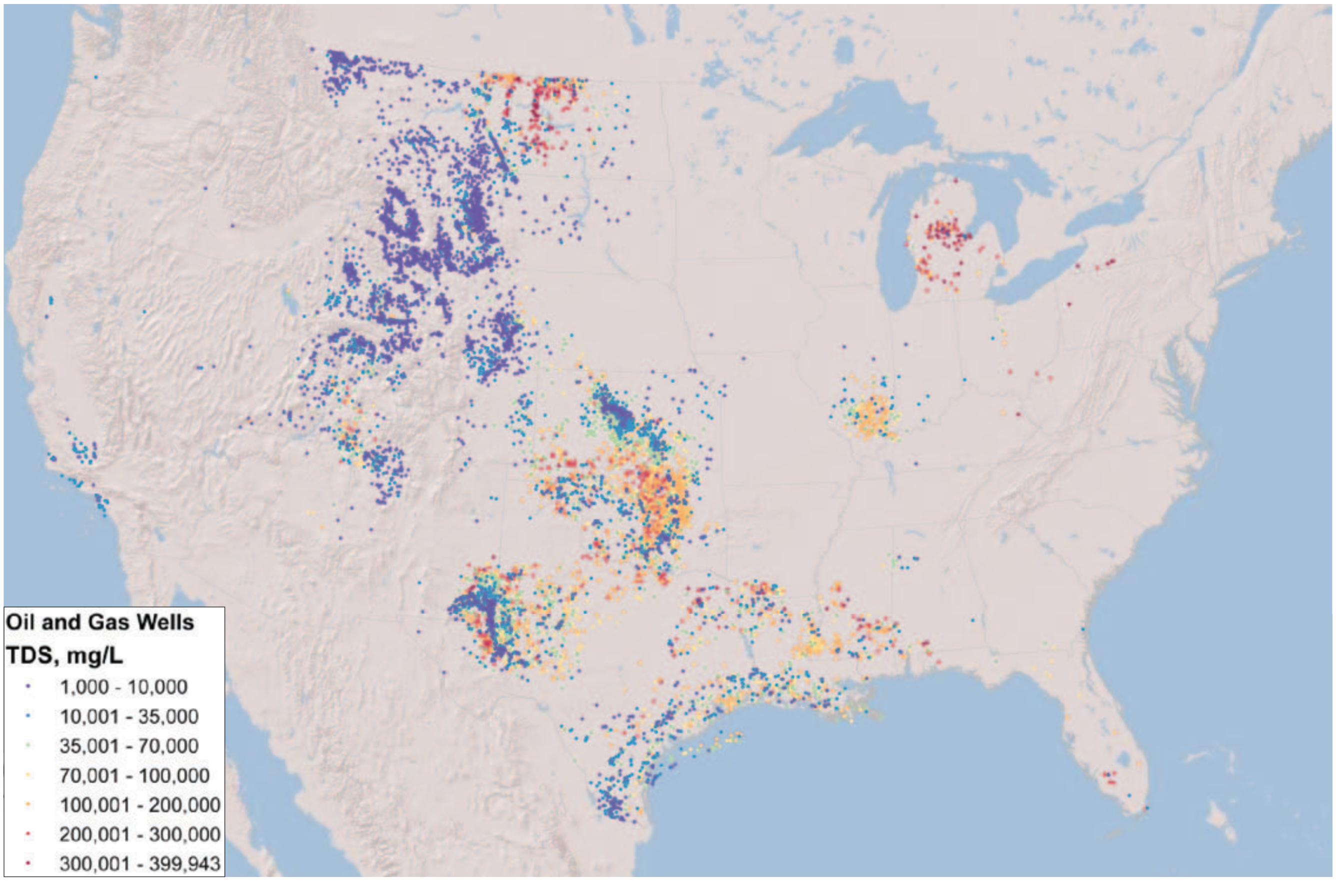
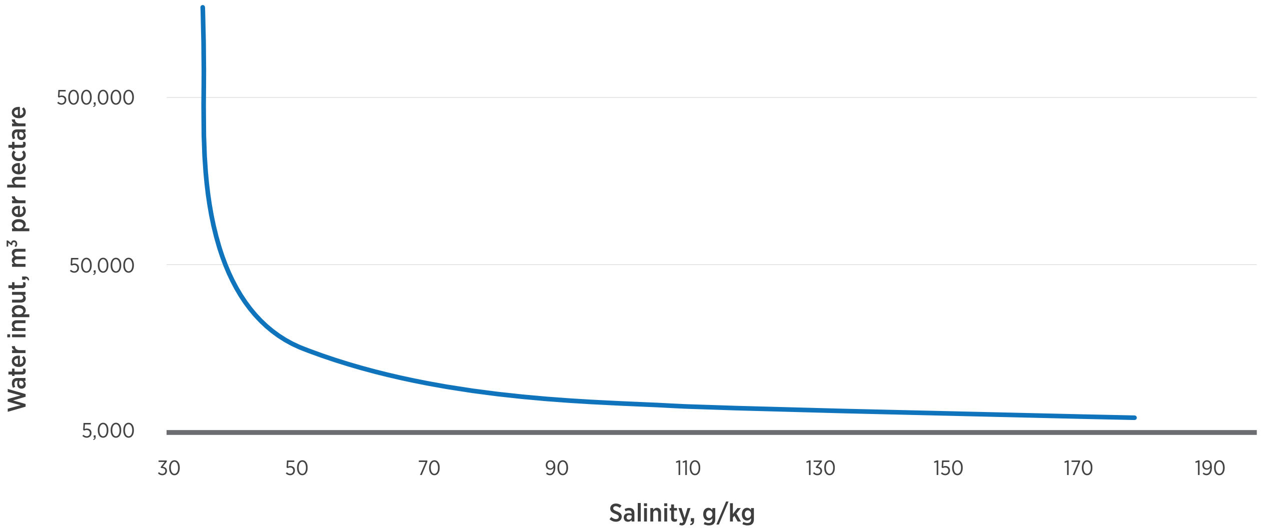


Figure 12.5. | Modeled relationship of water input to maintain a target open-pond salinity, considering salinity of the water source, blowdown and evaporative loss in Tucson, Arizona. Site-specific curves such as this are required for operational planning in order to maintain a target salinity for the cultivated algal strain.



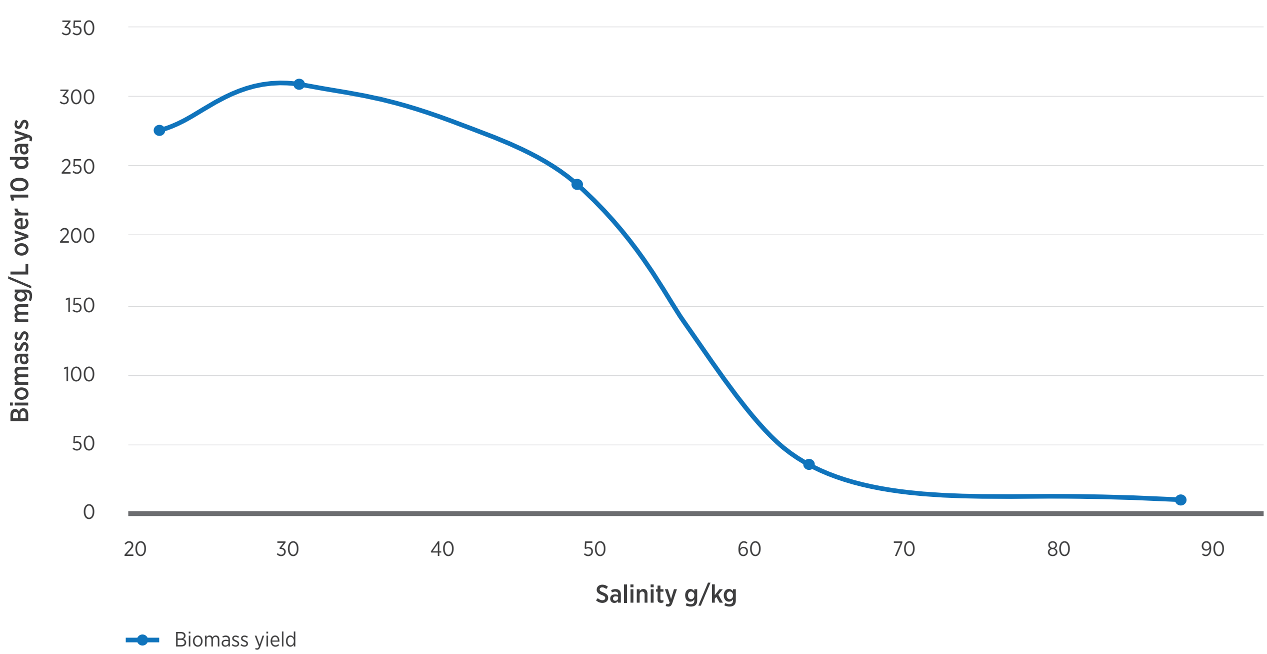
It should be noted that the exact form of the quantitative relationship shown in figure 12.5 is site-specific and varies depending on location and local climate conditions. Also, different strains of microalgae have varying tolerances for salinity concentrations and optimal ranges that provide opportunity for the highest growth rates. Figure 12.6 illustrates how careful management of saline algal strains is imperative; an optimal salinity of ~30 g/kg provides the highest productivity rate for *Nannochloropsis salina*, which then decreases with an increase in salinity concentration.

In Venteris et al. (2013), a water trade-off analysis is conducted that considers the ratio of algae-produced bio-oil to total consumptive water use required in cultivating the algal biomass; water availability of seawater, saline groundwater and freshwater; and the costs associated with delivering each type of water to potentially suitable open-pond sites identified in Wigmosta et al. (2011). In Venteris et al. (2013) potential sites were screened for cost-effectiveness targets for water delivery. Delivery could not exceed 20%, 10%, or 5% of a $2.90/gallon wholesale renewable diesel cost, targets which would represent the most water-efficient and cost-efficient sites within the conterminous United States. In the majority of cases, freshwater was the least-expensive source of water, but this resource is often subject to a high degree of competitive use. Saline groundwater was often the next most-economical water source, due to shorter transport distances than from seawater-based sources.

Saline versus Freshwater—Water Consumption

The CO_2 co-location scenarios developed in *BT16* volume 1 included consumptive water use simulations of freshwater and blowdown requirements for saline water as part of the physics-based mass and energy balance models that predict biomass growth (Wigmosta et al. 2011). For the freshwater scenarios, the consumptive water use was tracked as evaporative water loss from the open ponds and total loss from a unit farm. For the saline water scenarios, following the procedure in Venteris et al. (2013), the pond salinity was set to a concentration of 60 g/kg; as water evaporated, pond salinity increased, and thus, a given quantity of pond water was expelled (as blow-

Figure 12.6. | Relationship between biomass productivity and salinity concentration for *Nannochloropsis salina*



down), and new water was brought in to maintain the required salinity. The blowdown rates are variable by site, as the source water salinity and meteorology are site-specific. Water use for freshwater is focused on consumptive use (evaporative loss), and water use for saline water is more focused on the movement of and use of water for blowdown requirements.

In this chapter, consumptive water use for both saline water and freshwater sites co-located with existing waste CO_2 sources (ethanol production plants, coal power plants, natural gas power plants) are reported through the use of national maps, rather than by reporting quantitative totals for each site and scenario. The intent is to provide the reader with a qualitative understanding of consumptive use patterns under different scenarios and locations throughout the country. Each figure set is organized to illustrate peak consumptive water use (summer months, top figure) and annual average rates (bottom figure), which can differ significantly. The following scenarios are presented (table 12.4).

The most notable pattern across all scenarios is in the difference in water use between the average hourly summertime use and average hourly use across the year. This is particularly evident in the western states and, to a lesser extent, in the Midwest, whereas areas along the Gulf Coast, southeastern states, eastern seaboard, and Great Lakes regions have a smaller difference of water use seasonally. The regional and seasonal differences in consumptive use are driven by notable differences in the climate, where relative humidity plays a significant role in evaporative water loss.

The constraint on freshwater use had differing outcomes, depending on the co-location scenario. Overall, instituting the 5% mean annual flow rule for freshwater sites did not significantly impact the number of potential algal production sites available under the ethanol production and natural gas power plant co-location scenarios. In these cases, the physical co-location with an economically viable waste CO_2 source was the most significant limiting factor. However, because of the large number of coal-fired power plants available for co-location, under the freshwater scenario, potential sites were in fact limited as compared to sites that were sourced with saline water, which did not have explicit constraints on total use. This was most notable in the Gulf Coast region, southeastern states, and eastern seaboard.

For saline water, the source salinity constraints were broad in this analysis and as such, potential algal production sites were rarely excluded based on saline

Table 12.4. | List of Illustrative Algae Production Scenarios (combinations of variables) for Which National Consumptive Water Use Is Presented

Figure	CO_2 Co-Location Source	Water Source	Algal Strain
Figure 12.7	Ethanol	Saline	*Nannochloropsis salina*
Figure 12.8	Coal power plant	Saline	*Nannochloropsis salina*
Figure 12.9	Natural gas power plant	Saline	*Nannochloropsis salina*
Figure 12.10	Ethanol	Freshwater	*Chlorella sorokiniana*
Figure 12.11	Coal power plant	Freshwater	*Chlorella sorokiniana*
Figure 12.12	Natural gas power plant	Freshwater	*Chlorella sorokiniana*

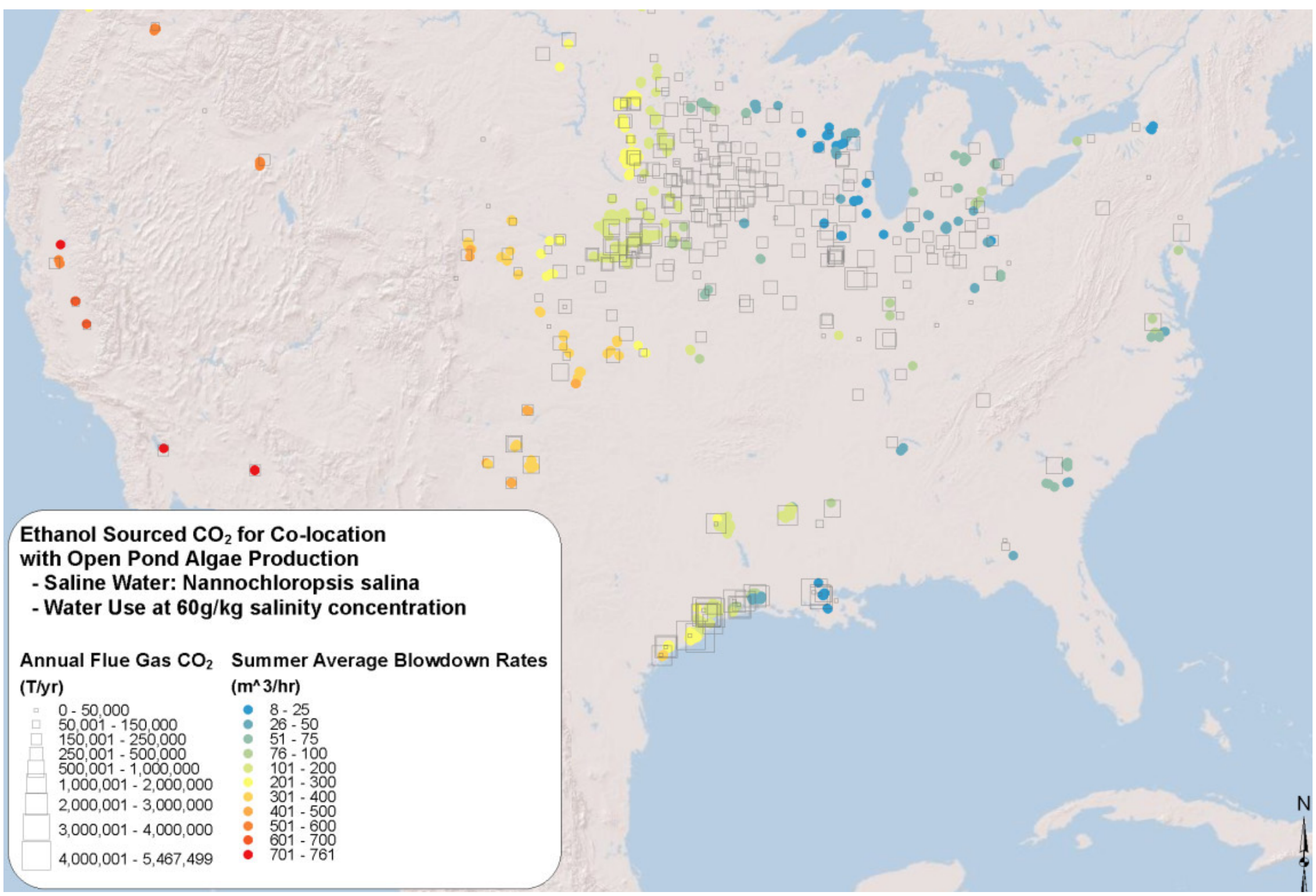
For each scenario above, both summer average hourly blowdown rates and annual average hourly blowdown rates are shown.

water availability, but as with freshwater, co-location to a waste CO_2 source was the primary siting constraint. Of importance with respect to freshwater are the seasonal water scarcity estimates illustrated in figure 12.7 (see also discussion in section 12.5.2.3, Water Consumption and Timing of Supply). Despite the use of the 5% mean annual water use rule, a number of sites across all co-location scenarios—but especially natural gas and coal power plant co-location scenarios—could potentially be impacted by high seasonal water scarcity, particularly under an altered climate. Note, however, that the water scarcity analysis is based on local water scarcity; thus, a major upstream water source (e.g., Missouri River, Mississippi River) could potentially dampen the water scarcity risk. Careful regional evaluation, long-range planning, competitive use, and climate-based risk evaluation, with respect to sustainable water use, is required. With regards to saline water, while best available public data and geostatistical analysis of these data were put forth, there is still a significant degree of uncertainty in the saline water estimates, partly due to uncertainties of geologic formations and high variability in ion chemistry (Venteris et al. 2013). In addition, site-specific assessments to sample saline waters for toxicants are necessary.

Productivity

Under the future productivity scenarios from *BT16* volume 1, a decrease in consumptive water use per unit biomass would be expected during the algae cultivation phase, as more biomass is produced in the same amount of time as under the current productivity scenarios. Increased productivity, however, also means more harvesting, dewatering, and pro-

Figure 12.7 | Blowdown rates for 405-ha saline open ponds co-located with ethanol production plants as average hourly summertime rate (*top*) and average hourly rate over the year (*bottom*)



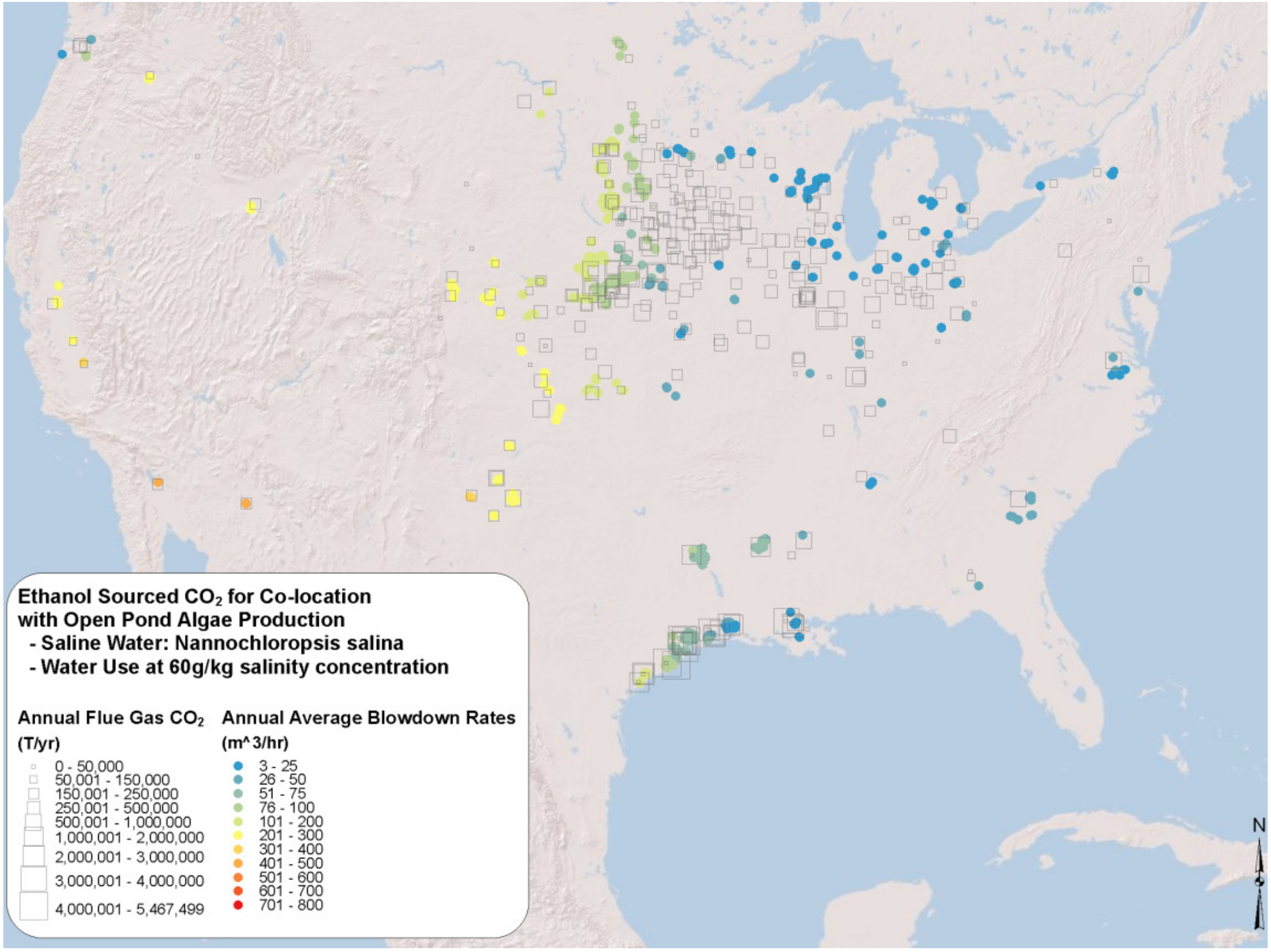

Ethanol Sourced CO_2 for Co-location
with Open Pond Algae Production
- Saline Water: Nannochloropsis salina
- Water Use at 60g/kg salinity concentration
Annual Flue Gas CO_2
(T/yr)
0 - 50,000
50,001 - 150,000
150,001 - 250,000
250,001 - 500,000
500,001 - 1,000,000
1,000,001 - 2,000,000
2,000,001 - 3,000,000
3,000,001 - 4,000,000
4,000,001 - 5,467,499
Annual Average Blowdown Rates
(m^3/hr)
3 - 25
26 - 50
51 - 75
76 - 100
101 - 200
201 - 300
301 - 400
401 - 500
501 - 600
601 - 700
701 - 800
N

Figure 12.8 | Blowdown rates for 405-ha saline open ponds co-located with coal-based power plants as average hourly summertime rate (*top*) and average hourly rate over the year (*bottom*)

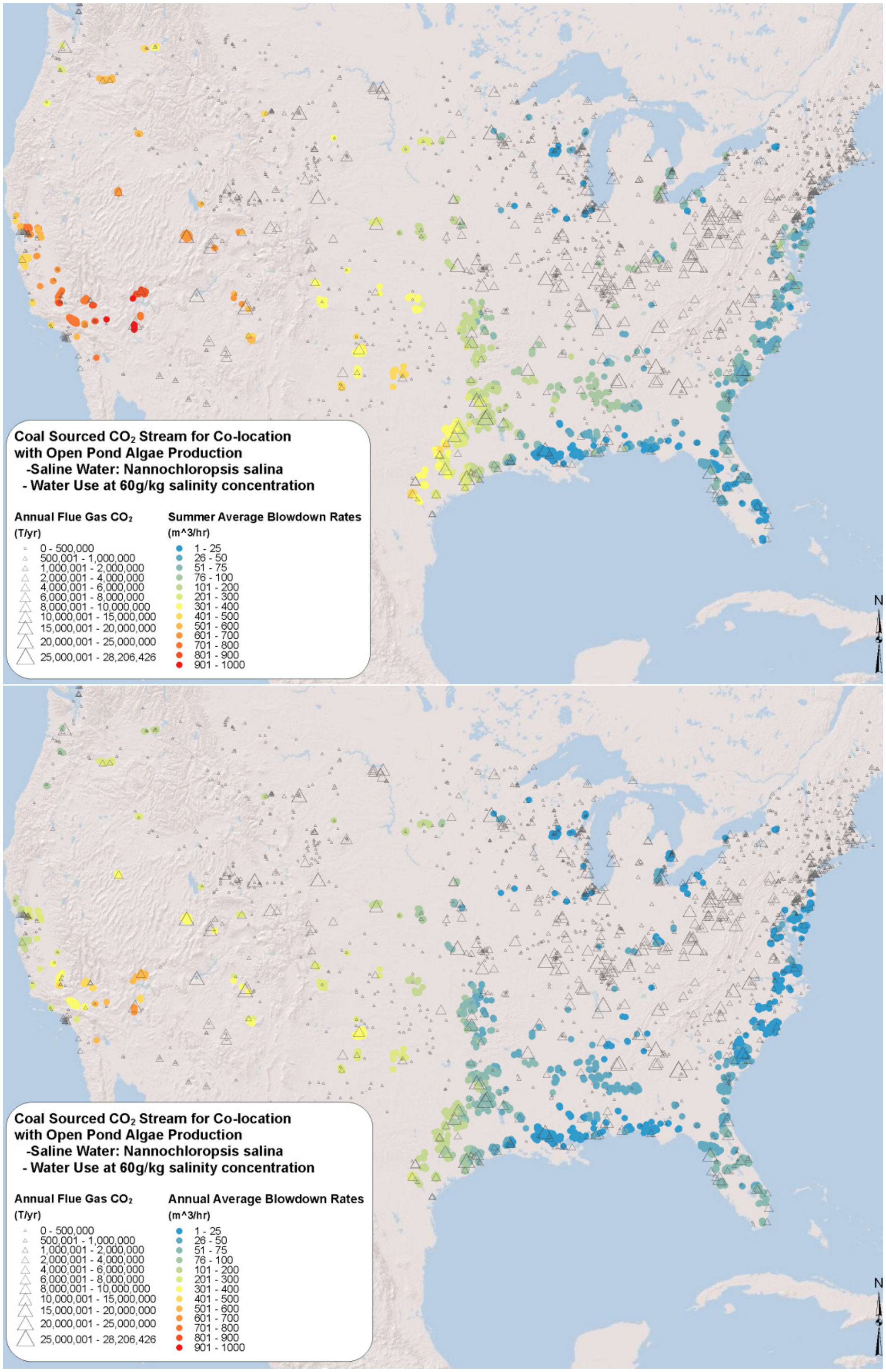


Figure 12.9 | Blowdown rates for 405-ha saline open ponds co-located with natural gas–based power plants as average hourly summertime rate (*top*) and average hourly rate over the year (*bottom*)

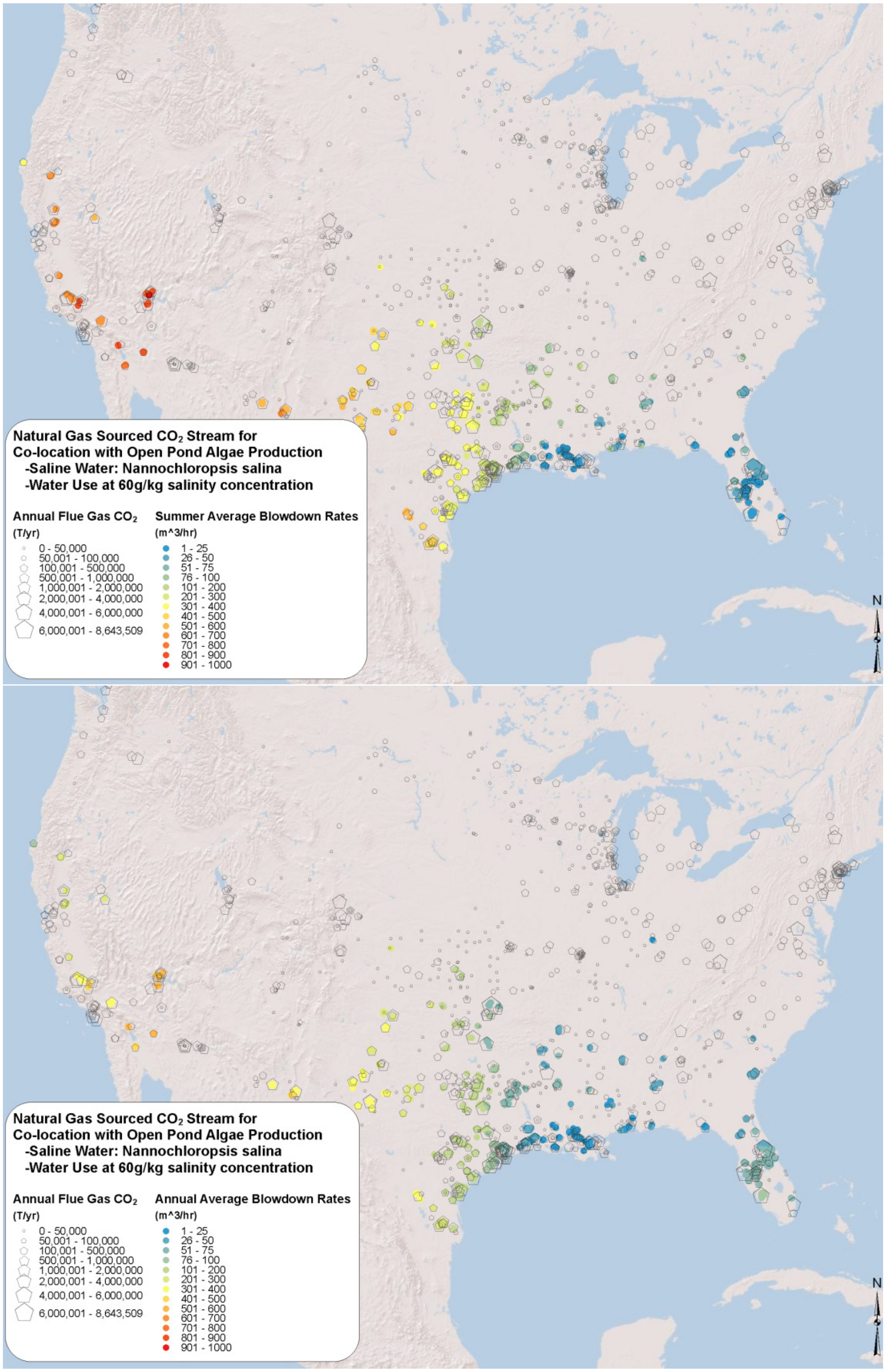
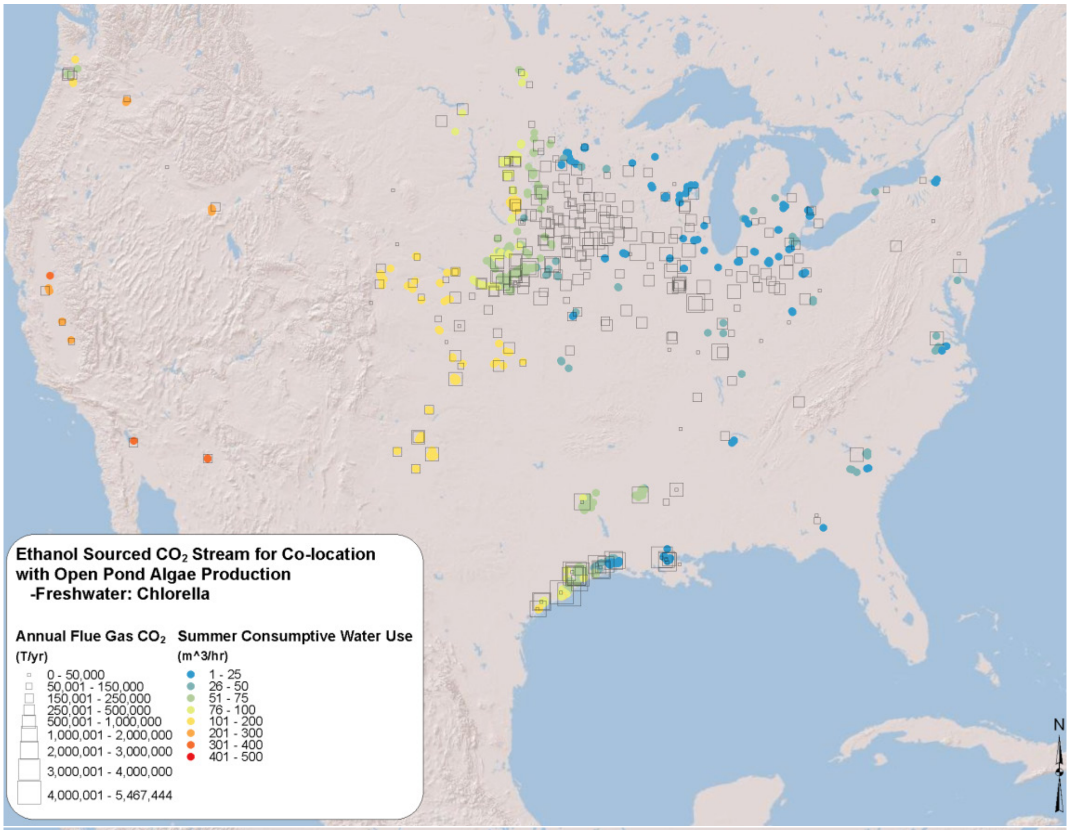


Figure 12.10 | Consumptive freshwater use for 405-ha freshwater open ponds co-located with ethanol production plants as average hourly summertime rate (*top*) and average hourly rate over the year (*bottom*)



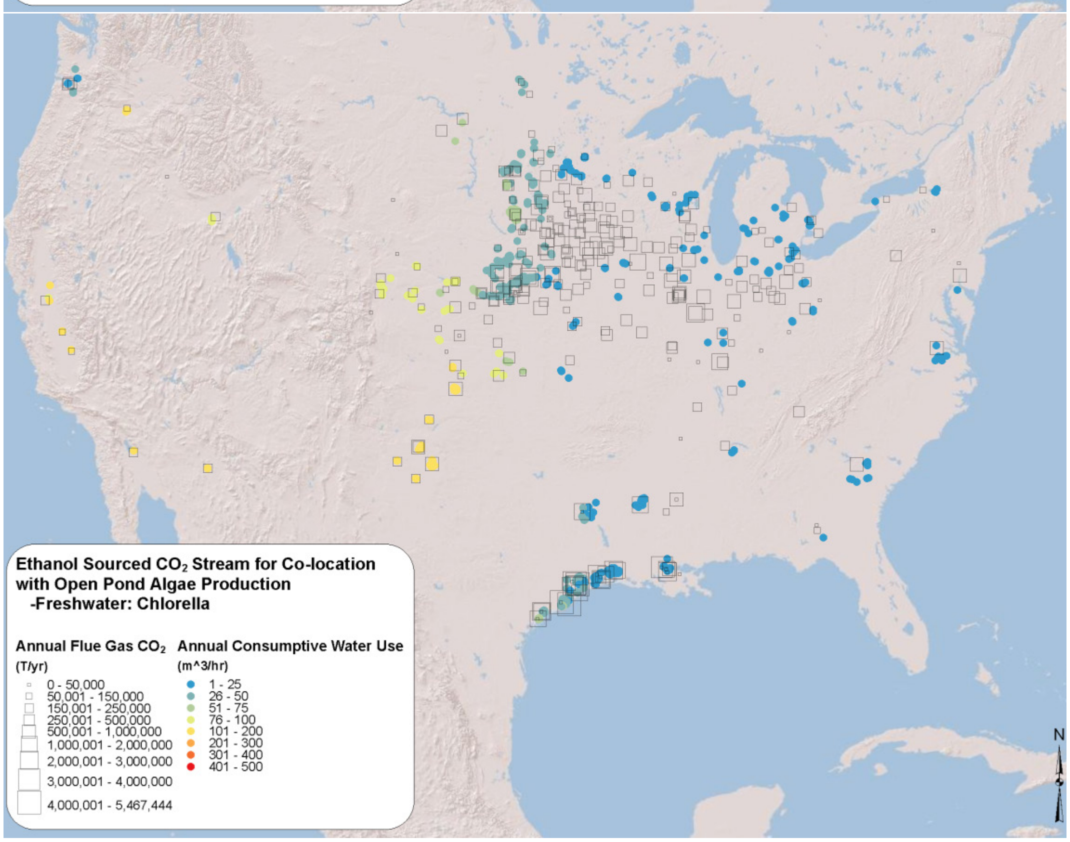


Figure 12.11 | Consumptive freshwater use for 405-ha freshwater open ponds co-located with coal-based power plants as average hourly summertime rate (*top*) and average hourly rate over the year (*bottom*)

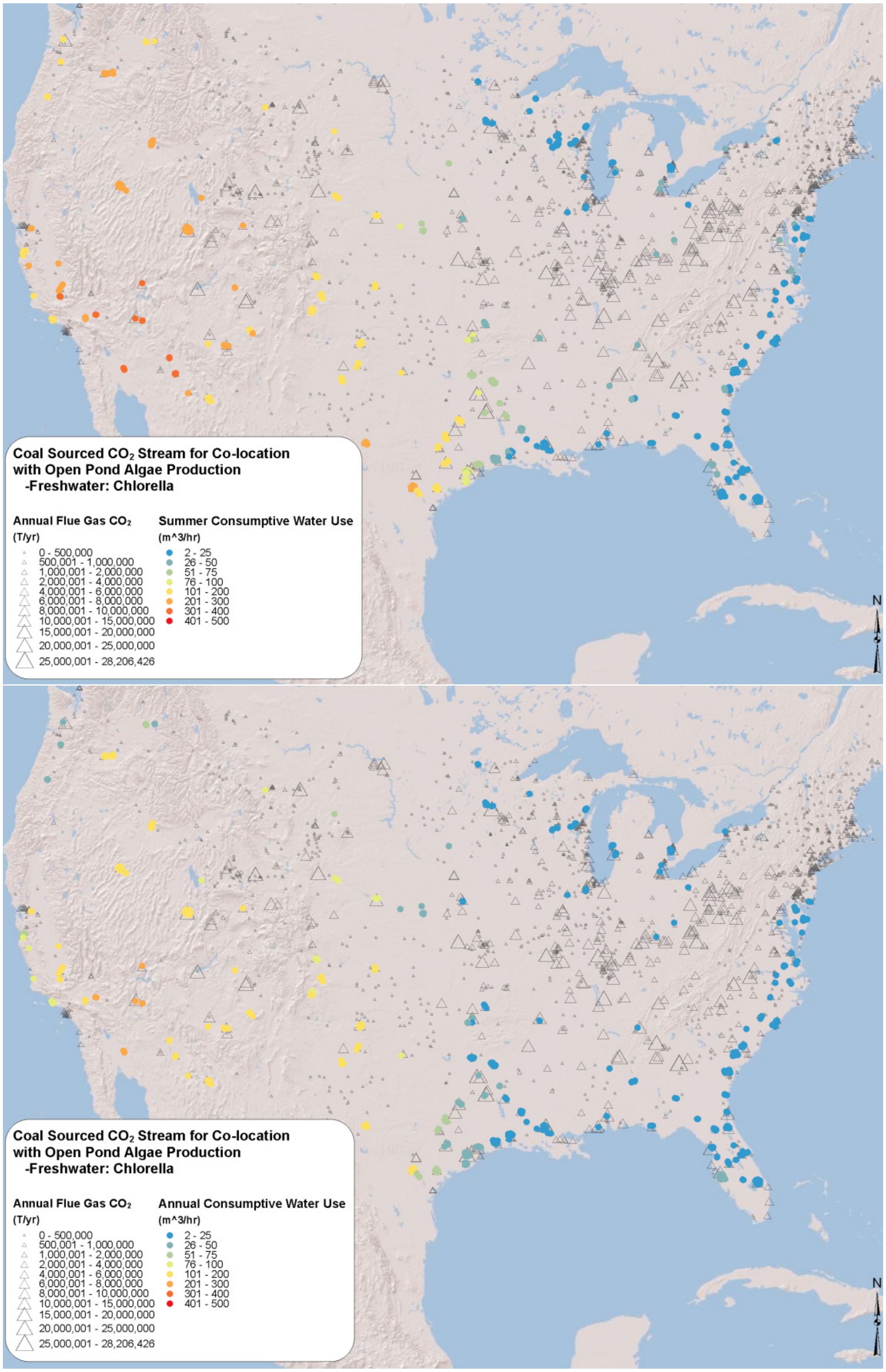
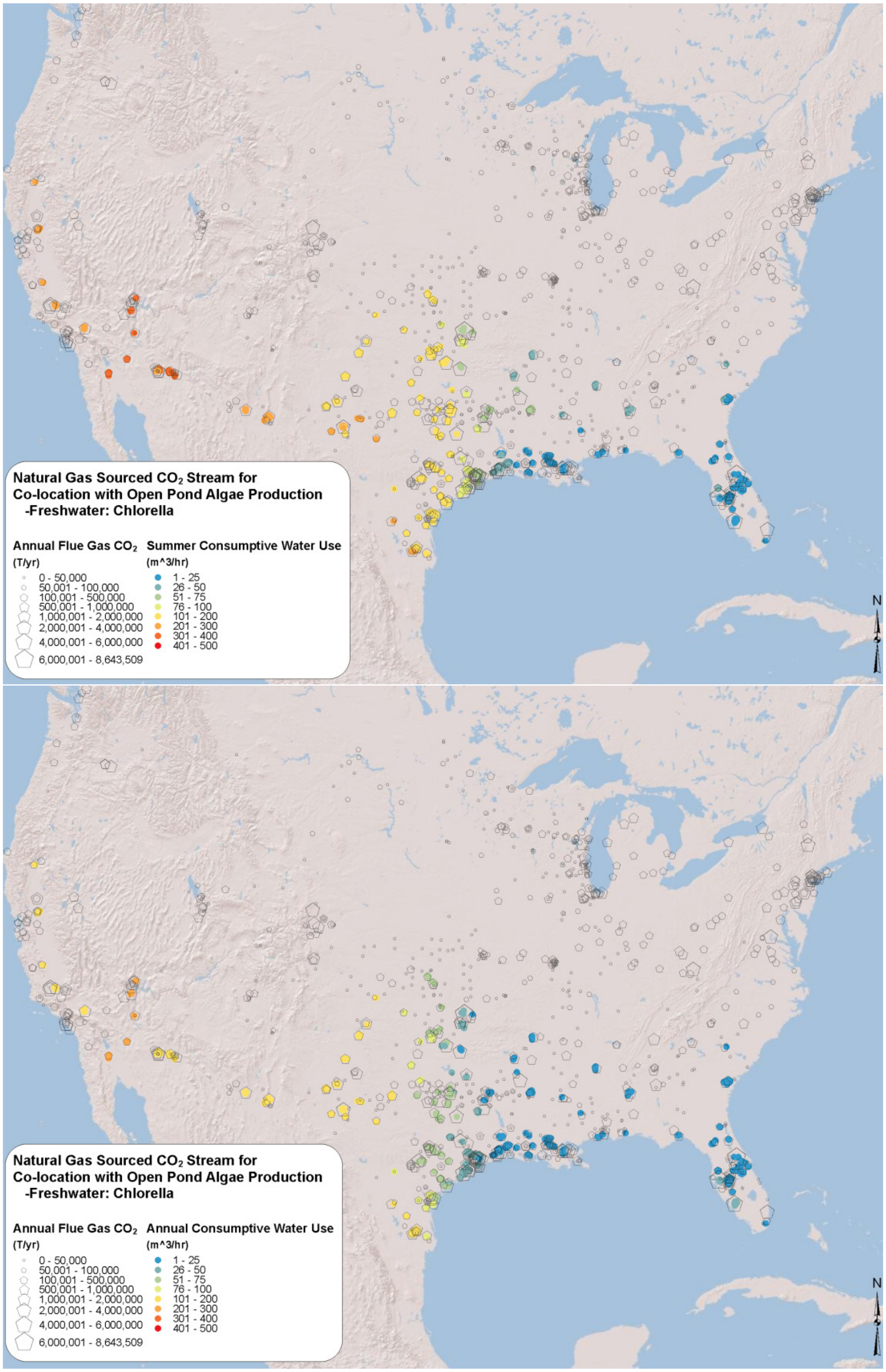


Figure 12.12 | Consumptive freshwater use for 405-ha freshwater open ponds co-located with natural gas power plants as average hourly summertime rate (*top*) and average hourly rate over the year (*bottom*)



cessing—thus, potentially driving more water losses through the system per unit of time. This largely depends on how a site or an enterprise of sites is configured and whether or not harvested biomass is processed on-site, within the enterprise, or moved at distance, creating “virtual water” (water embedded in a product and exported outside of the basin of origin) and likely allowing limited opportunity for water recycling.

Full Plastic Liners versus Minimal Plastic Liner

The extent of a pond liner may have implications for water quantity and quality. Many commercial- and research-scale algae-cultivation facilities install plastic or clay liners beneath cultivation ponds to prevent or mitigate water loss due to seepage, as well as to control the release of salts and nutrients into the subsurface or groundwater. Lined ponds include plastic liners (e.g., HDPE) that create barriers with very low effective permeability ($<10^{-12}$ cm/s), high longevity, and resistance to chemical and ultraviolet light degradation (Ng 2008).

Yet, liners are expensive (see chapter 7, volume 1 of *BT16*), and are not always needed. Venteris et al. (2014) proposed that natural soil properties, particularly soils with low saturated hydraulic conductivity (Ks), could be used to avoid the costly installation of plastic liners.

Evidence from the laboratory and field illustrates that liners are not always needed if the objective is to avoid leaching of cultivation fluids. The DOE-funded *Aquatic Species Project* tested unlined ponds for 2 years at a site in New Mexico without observable leakage (Brown and Sprague 1992). Studies of animal waste settling ponds and related industrial ponds show that underlying soils can exhibit reduced hydraulic conductivity over time, reducing or eliminating the need for HDPE or clay liners (SNTC 1993). Numerous lab-scale and field-scale studies have demonstrated rapid development of low-conductivity seals in soils beneath animal waste settling ponds (Cihan et al. 2006; Culley and Phillips 1982; Barrington et al. 1987a, 1987b; Rowsell 1985; Hills 1976; Chang et al. 1974). The mechanisms are physical and microbial pore clogging that may occur due to rearrangement of soil particles and growth of microbial biomass and buildup of metabolic products (Barrington et al. 1988; Barrington et al. 1987a, 1987b; Chang et al. 1974).

Vandevivere and Baveye (1992) established that various strains of bacteria differ in growth and metabolic-production rates, resulting in different degrees of decreased hydraulic conductivity of porous media. Numerous studies since the late 1940s indicate two to three order-of-magnitude reductions of saturated hydraulic conductivity can be credited to growth of such bacteria (Thullner 2010).

Soils ranging from commercial-grade “play sand” to clay loam soils develop seals that readily converge to a similar hydraulic conductivity (Cihan et al. 2006; Barrington et al. 1988; Barrington et al. 1987a, 1987b; Cihan et al. 2006) within days to weeks after construction. The soil-plugging process may be insensitive to soil texture (Hills 1976; Culley and Phillips 1982; Rowsell et al. 1985; Barrington et al. 1987a, 1987b). Thus, site discrimination according to hydraulic conductivity (Venteris et al. 2014) may be unnecessary (i.e., many soils can exhibit reduced hydraulic conductivity at $<10^{-7}$ cm/s).

For soil liners containing most types of wastes, hydraulic conductivities of $1x10^{-7}$ cm/s are required to prevent leaking of nutrients or contaminants into the subsurface (Daniel and Benson 1990). Best management practices are needed to achieve this water quantity (and water quality) target.

Research is needed to develop a mechanistic understanding of the processes that seal soils. Research is also needed to provide an experimental basis for understanding the characteristics of soils that will and will not seal sufficiently to allow unlined or minimally lined ponds.

12.4.2.3 Water Consumption in the Context of Agricultural Crops

Water consumption values need to be considered in the context of competing uses and regional availability. Here we consider the water requirements of agricultural crops.

A method of comparing water use between traditional agricultural crops and cultivated microalgae is made by considering the water use per mass of crop yield, i.e., the water footprint, consistently across the United States. Both precipitation or rainfall (sometimes termed "green water") and water withdrawn from surface and/or groundwater sources (sometimes termed "blue water") are presented for 11 terrestrial crops, varying from oil seed crops, to grains, to nuts, to the modeled freshwater algal strain, *Chlorella sorokiniana*. The water footprint for microalgae captures long-term annual total evaporative loss from the modeled 30-cm deep open pond and assumes 85% of pond water removed during harvest is recycled back to the pond. Methods and assumptions for modeling algae production are described above. Water use data were converted from units of L/ha to m^3/ha, and modeled biomass was converted from kg/ha to tons/ha to allow for comparison with terrestrial crops.

The water use data for terrestrial crops are sourced from Mekonnen and Hoekstra (2011) and are derived using a grid-based soil-water balance model and calculation of crop- and location-specific evapotranspiration (ET). The total annual water use is divided by the total annual crop yield to achieve a common water volume per mass produced (m^3 'withdrawn surface and/or groundwater' + 'direct precipitation'/ton of harvested crop/year)[5] that is averaged within a state-level boundary. ET is calculated daily using crop coefficients throughout the growing season considering available soil moisture in the rooting zone, plant growth stage, and meteorology (see Allen et al. 1998; Chapagain and Hoekstra 2004). For this comparison, we are considering crop yield and not necessarily the whole biomass produced. This provides a common unit across crops, since for oil seed crops, only the seeds are used, and for tree nuts, only the nuts are harvested; however, for algal biomass, the whole biomass is harvested. Results are reported in the following units: m^3 'withdrawn surface and/or groundwater' + 'direct precipitation' / ton crop yield. Table 12.5 provides a state-level assessment of annual average 'withdrawn surface and/or groundwater' + 'direct precipitation' crop water use by state for 11 common terrestrial crops and freshwater microalgae. Appendix 12-D provides state-level maps to visualize these results.

With the assumptions in the analysis, such as water recycle, microalgae consumptive water use per biomass yield ranges from 5–953 m^3/ton, which is favorable in comparison to several other crops, where consumptive water use per crop yield is generally higher across the U.S., such as sunflower (2,615–4,265 m^3/ton), rye (2,041–4,265 m^3/ton), and rapeseed/canola (519–2,899 m^3/ton). Further analysis of water use would consider a selection of bioenergy-potential crops where the whole biomass is used and converted to an end product (ethanol or biodiesel) where the water consumption per energy unit can be assessed. Algae may be a high-quality source of proteins. Water consumption for algae may be more favorable if effects of food and fuel are considered together.

12.4.2.4 Water Consumption and Timing of Supply

The consideration of water-resource availability with respect to timing of supply and demand is important for evaluating competitive use. Many water-scarcity and water-stress indices (see below) are measured at the annual scale, which overlooks the critical monthly-to-seasonal aspects of the systems, particularly regarding the large allocation of consumptive water use to the agricultural sector during the growing

[5] Withdrawn surface and/or groundwater in this study is equivalent to blue water; direct precipitation in this study is equivalent to green water in Mekonnen and Hoekstra (2011) and other studies.

Table 12.5. | Annual Average Green Water (Direct Precipitation) + Blue Water (Withdrawn Surface and/or Groundwater) Crop Consumptive Water Use by State for 11 Common Terrestrial Crops and 1 Freshwater Microalgae Crop (*Chlorella sorokiniana*)

Green + Blue Water Use (in m^3 water/ton of crop yield)												
State	Corn	Sugarbeet	Soybean	Chickpea	Almonds	Walnuts	Rapeseed/ Canola	Sunflower	Wheat	Rye	Sorghum	Microalgae-Chlorella
Alabama	647	-	1,879	2,086	1,097	-	1,879	-	1,225	3,226	1,097	32
Arizona	851	536	-	-	1,843	-	536	-	1,247	-	1,307	585
Arkansas	617	-	1,940	1,971	1,119	-	1,940	-	1,566	-	1,119	49
California	579	519	-	2,743	1,535	3,419	519	4,275	1,248	2,589	1,015	552
Colorado	734	905	1,994	2,354	2,078	-	2,899	3,889	2,929	3,771	1,173	661
Connecticut	462	-	1,215	1,625	-	-	1,215	-	-	2,699	-	-
Delaware	512	-	1,472	1,834	752	-	1,472	-	1,404	2,886	752	26
Florida	563	-	1,805	2,125	-	-	1,805	-	1,198	3,236	-	35
Georgia	576	-	1,724	2,016	908	-	1,724	-	1,368	3,150	908	32
Idaho	951	915	-	1,952	915	3,660	915	-	2,284	-	-	692
Illinois	584	-	1,618	1,887	870	3,123	1,618	3,289	1,449	3,014	870	39
Indiana	533	-	1,604	1,833	843	3,038	1,604	-	1,375	2,927	843	35
Iowa	554	-	1,617	1,885	948	3,031	1,617	2,970	1,546	2,968	948	83
Kansas	744	-	1,815	-	1,100	-	1,815	4,023	2,119	4,070	1,100	333
Kentucky	584	-	1,602	1,783	915	3,421	1,602	-	1,383	-	915	37
Louisiana	643	-	1,915	2,334	1,047	-	1,915	-	1,367	-	1,047	27
Maine	377	-	949	1,312	-	-	949	-	1,031	2,041	-	14
Maryland	518	-	1,515	1,784	775	3,107	1,515	-	1,471	2,945	775	30
Massachu-setts	457	-	1,220	-	-	-	1,220	-	1,318	2,751	-	26
Michigan	480	535	1,475	1,515	535	2,621	2,010	-	1,394	2,585	-	36
Minnesota	529	576	1,575	1,769	576	-	2,151	2,867	1,216	2,783	-	133
Mississippi	589	-	1,920	2,094	1,033	-	1,920	-	1,473	-	1,033	34
Missouri	618	-	1,678	1,970	992	3,304	1,678	3,579	1,474	2,162	992	51

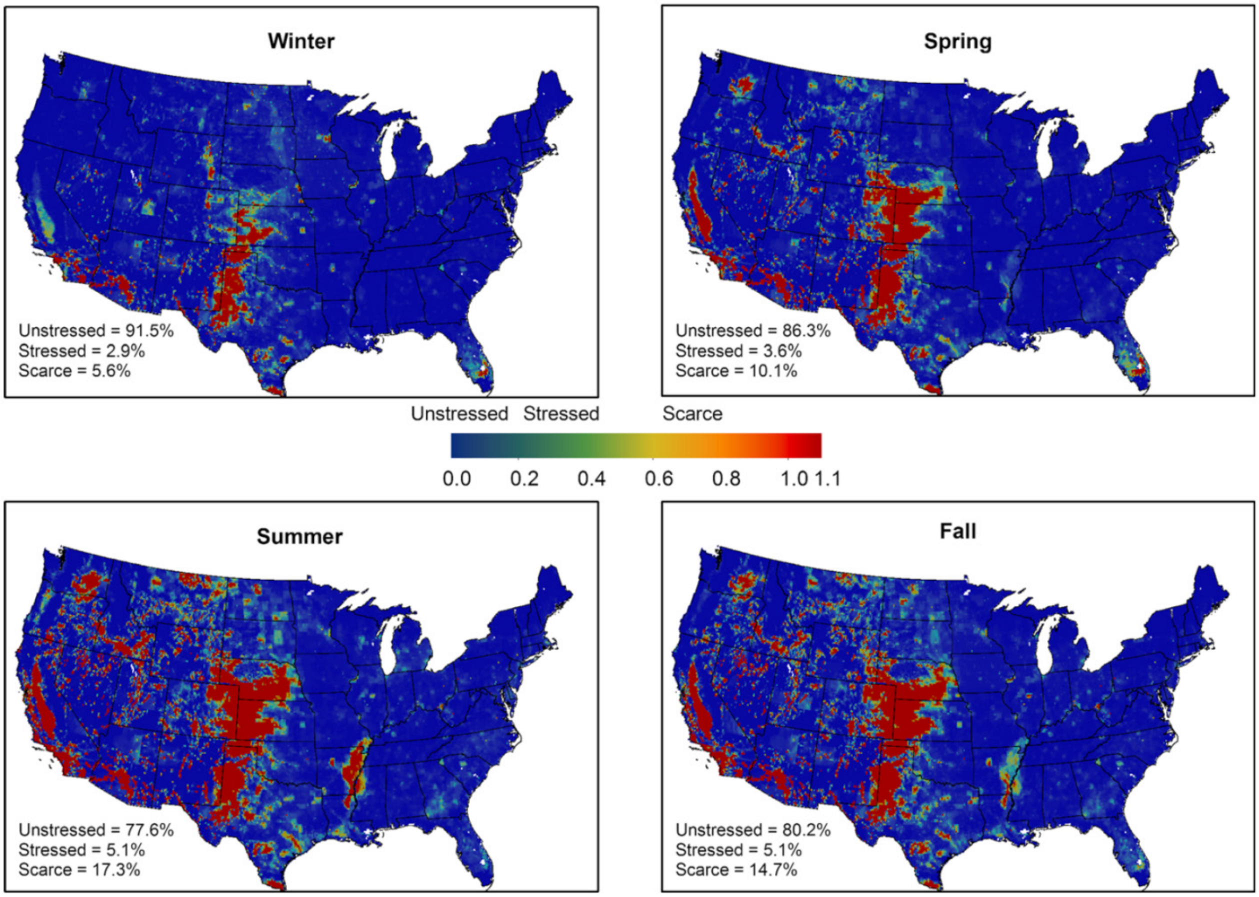
Green + Blue Water Use (in m³ water/ton of crop yield)												
State	Corn	Sugarbeet	Soybean	Chickpea	Almonds	Walnuts	Rapeseed/ Canola	Sunflower	Wheat	Rye	Sorghum	Microalgae-Chlorella
Montana	596	772	-	1,959	772	-	772	3,386	2,406	3,170	-	623
Nebraska	634	852	1,802	2,127	1,895	3,207	2,654	3,696	2,316	3,403	1,043	289
Nevada	885	-	-	2,513	-	-	-	-	2,659	-	-	923
New Hampshire	394	-	-	1,164	-	-	-	-	-	-	-	14
New Jersey	449	-	1,377	1,773	762	-	1,377	-	1,663	2,929	762	19
New Mexico	793	-	1,830	3,264	1,177	-	1,830	4,046	2,618	4,115	1,177	567
New York	449	-	1,358	1,565	-	-	1,358	2,683	1,373	2,550	-	30
North Caro-lina	532	-	1,600	-	730	3,071	1,600	-	1,426	2,783	730	16
North Dakota	532	597	1,569	1,749	597	-	2,166	2,999	1,257	2,935	-	323
Ohio	493	573	1,495	1,706	1,397	2,799	2,068	-	1,413	2,692	824	23
Oklahoma	771	-	2,224	2,819	1,176	-	2,224	3,845	2,308	4,265	1,176	282
Oregon	690	1,009	-	2,350	1,009	2,702	1,009	-	2,342	3,092	-	662
Pennsylvania	457	-	1,466	1,610	728	2,721	1,466	2,797	1,603	2,742	728	5
Rhode Island	470	-	-	1,599	-	-	-	-	-	2,686	-	-
South Car-olina	549	-	1,701	2,013	760	-	1,701	-	1,336	3,021	760	28
South Dakota	603	-	1,599	1,919	958	3,156	1,599	3,400	1,854	3,129	958	321
Tennessee	652	-	1,708	-	989	-	1,708	-	1,459	3,091	989	23
Texas	810	-	2,002	3,039	1,206	-	2,002	4,276	2,339	2,723	1,206	298
Utah	746	860	-	3,285	860	3,909	860	-	2,608	-	-	863
Vermont	412	-	1,137	-	-	-	1,137	-	1,229	2,317	-	36
Virginia	529	-	1,614	1,869	764	3,052	1,614	-	1,469	2,796	764	28
Washington	636	922	-	2,056	922	3,144	922	-	2,390	-	-	372
West Virginia	442	-	1,450	1,656	-	2,952	1,450	-	1,507	2,694	-	-
Wisconsin	468	-	1,438	1,771	-	2,675	1,438	2,615	1,517	2,578	-	19
Wyoming	656	791	-	2,090	1,792	-	791	3,641	2,902	-	1,001	953

Terrestrial crop water use sourced from Mekonnen and Hoekstra (2011); microalgae water use sourced from Wigmosta et al. (2011)

season. The use of these indices as environmental metrics presents a challenge for evaluating competing water use with respect to algae production, for which the summer months generally have a higher consumptive water use. There are regional exceptions to high summer crop irrigation requirements; for example, in Florida, summer precipitation negates the need for crop irrigation, and the nature of the dominant crops (citrus) requires more irrigation in the fall, winter, and spring. Also, for algae production, higher levels of humidity in the summer months reduce the evaporation loss from open ponds, compared to loss in other seasons.

To highlight the seasonal changes, Moore et al. (2015) calculated local water scarcity based on the water scarcity and the classification method of Sun et al. (2008a, 2008b) (see fig. 12.13), where the following categorizations are defined: <0.2=unstressed; ≥0.2–<0.4=stressed; and ≥0.4=scarce.[6] In the Midwest, the Ogallala Aquifer is consistently in a water-scarce classification, largely due to limited precipitation and large agricultural water use (primarily due to groundwater pumping). Many agricultural regions in the western United States and, notably, the Mississippi River Plain move to a water-scarce condition during the growing season. The cultivation of algal biomass would have to take seasonal water requirements for competitive uses of water into account for siting and planning.

Figure 12.13. Seasonal local water scarcity for the conterminous United States. The indicated percentages of unstressed, stressed, and scarce reflect the fraction of 1/8° cells that fall in one of these three categories (Image credit: Moore et al. 2015).



6 Although Sun et al. (2008a, 2008b) term their method a "water stress index," it is a water scarcity index by the UN definition above, since it considers volumetric supply of water and not water quality, accessibility, and environmental requirements.

12.4.2.5 Water Consumption and the Supply Chain

This assessment herein has focused on the cultivation aspects of microalgae production and not on the water aspects through the full supply chain. This has partly been due to the limits of the study in producing algal biomass to the so-called ‘pond gate’ and not considering the various downstream extraction, processing, and fuel upgrading pathways for the feedstock. Additional analysis with considerations and variants from the cultivation process and operations through to fuel products is required to shape the water considerations in the full supply chain.

The differences in water use can vary significantly starting with the cultivation operations where open pond operations are site-specific and specific to environmental conditions and may or may not include pond water recycle, and may harvest at different concentrations where water is lost with each harvest and may or may not be recycled back into the system. Beyond the harvesting, different technological approaches to dewatering are further dependent on the downstream processing and product end point.

The use of algae for fuels can involve a lipid-only extraction. Alternatively, a whole algal biomass slurry can be processed into a bio-oil intermediate through hydrothermal liquefaction (HTL), after which oil is upgraded and fractionated into a variety of end fuel products. The HTL process can recycle most of the water that is put into the system. The fuel upgrading, whether by an HTL bio-oil or algae lipid, requires water for the refining process (i.e., cooling water, boiler, steam, backwash/rinse) (Wu et al. 2009; Luo et al. 2010). Design considerations as to whether lipid extraction or HTL processing are occurring on-site or are transferred elsewhere can have an impact of total site water use and feasibility for water recycle.

The use of PBRs will potentially minimize consumptive water use due to evaporative loss, but spray cooling is sometimes used to control the temperature of the media (NRC 2012). Algae-based ethanol secretion methods will also have a different consumptive water use where water losses are minimized in these closed systems. Direct ethanol secretion avoids a dewatering step and continually maintains a live algae culture.

12.4.3 Water Quality

Water quality of effluents from algal-biofuel facilities and receiving waters is influenced by the source of the water, nutrients and other amendments, and by the efficiency of nutrient use. Water quality of natural groundwater or surface water outside of the cultivation system is important for regional environmental objectives.

Nutrient-related indicators are important metrics of water quality for all biomass production, including algae (table 12.1). The net effect of algae cultivation systems on water quality depends on the baseline land management system, the quality of water released to natural environments, and the use of algae to modify water quality of associated systems, such as wastewater treatment. Eutrophication is caused or exacerbated by runoff from traditional agricultural systems; algae cultivation systems that produce a protein coproduct could replace other agricultural systems.[7] Overtopping or slow leakage of cultivation media from ponds to groundwater or surface water may occur in many ecosystems. (The risk would be lower for PBRs.)

If treated wastewater from wastewater treatment or concentrated animal feeding operations is used as a co-located nutrient source (not considered in the *BT16* volume 1 scenarios), downstream concentrations of nutrients in streams, as well as freshwater needs, should be positively affected by algae cultivation. As Chiu and Wu (2013) note, between 3% and 91% of water at the state level could be displaced by municipal wastewater. However, the risks to productivity from variable water chemistry and added

[7] In the scenarios discussed, however, croplands are not used for algae production.

microbes from some wastewater have yet to be overcome at a large scale (Shurin et al. 2013).

Downstream process elements could have a positive effect on water quality. Recycling of nutrients and algae would reduce nutrient loadings to streams (Murphy and Allen 2011). On-site water treatment would also reduce nutrient concentrations in effluents.

The use of impure gases from co-located power plants (coal-fired power plants and natural gas plants in volume 1 scenarios) could increase metal-contaminant loads in cultures and, ultimately, in natural waters. The potential accumulation of flue-gas-related contaminants in cultivation systems is not well understood but is beginning to be investigated. Examples include the incorporation of metals from coal-based flue gas in *Scenedesmus obliquus* (Napan et al. 2015) and *Desmodesmus communis* (Palanisami et al. 2015). Metal concentrations were not recommended as a generic indicator of water quality for algae biofuel systems in Efroymson and Dale (2015), but if algae affect levels of metals in surface or groundwater, then metal concentrations could be employed as an environmental indicator.

Because algae may be grown in coastal waters or saline or brackish groundwater, salinity of groundwater or surface water will sometimes be an important environmental indicator (table 12.1), as recommended by the NRC (2012). Unintentional leakage from open ponds, withdrawal effects, or injection of saline waste into the ground could lead to the possible salinization of groundwater or surface water in some environments. However, such salinization is hypothetical and has not been demonstrated. Water-quality effects could result from the construction and operation of pipelines to transport coastal waters to inland cultivation systems.

Water quality effects of ponds with plastic liners versus minimal or no plastic liners are described in the previous section on water quantity. Essentially, if permeability is very low (conductivity $<10^{-7}$ cm/s) due to physical and microbial clogging, then unlined ponds should not leach water or nutrients into underlying soils. If hydraulic conductivity is higher, adverse effects on groundwater quality could occur.

12.4.4 Other Environmental Indicators

12.4.4.1 Soil Quality

Soil quality is an important aspect of environmental effects of terrestrial biomass crops, which draw nutrients from the soil. Unlike vascular plants, algae do not extract nutrients or water from local soil, so soil nitrogen and phosphorus have not been proposed as environmental indicators for algae (table 12.1), as they are for terrestrial crops. Soil quality affects productivity of vascular bioenergy crops and ecosystems but not algae used for biofuels.

The main linkages of algal biomass production to soil quality are via excavation for construction and ultimate decommissioning. Therefore, bulk density has been proposed as an environmental indicator for algal biomass (table 12.1). If construction of ponds is performed at commercial scale, the top layers of soil may be compacted or removed (Davis et al. 2016), affecting soil density, potentially affecting soil carbon, and potentially creating a barrier between the surface soil and subsoil.

If unlined or partially lined ponds are used, soil nutrients are more likely to be affected by algae biomass production. With respect to the full supply chain, if residual algal biomass is used as fertilizer or a soil amendment, it has the potential to provide benefits to soil quality, particularly carbon and nitrogen.

12.4.4.2 Biodiversity

Algal biofuel production could affect aquatic or terrestrial biodiversity, but little research exists to support hypotheses related to algal biomass and biodiversity outside of the cultivation system. It is reasonable to assume that extensive freshwater or saltwater ponds across the landscape could affect populations of vertebrates. If wildlife were to drink from algal biomass ponds, potential toxic exposures to individuals could come from metals accumulated from flue gas, salinity, or toxins from opportunistic cyanobacteria (Kotut et al. 2010). Moreover, the high productivity of algae per acre, combined with the potentially large yields of protein coproducts, could result in decreased pressure for deforestation (and decreased pressure on forest biodiversity), compared that which could be associated with terrestrial crops.

Breaches or overtopping events could lead to large quantities of algae and nutrients released to aquatic ecosystems, causing some algal taxa to bloom, and potentially causing changes in the native community. However, it is reasonable to assume that existing or future best management practices would prevent or lower the risk of these events. Algae biomass production in marine waters is not considered in this study, and potential implications of production in coastal ecosystems have not been studied.

The selection and interpretation of biodiversity indicators should be specific to the region where they are applied. Therefore, the indicators' "presence of taxa of special concern" and "habitat area of taxa of special concern" (table 12.1) would be regional.

None of the variables from the scenarios in volume 1 are directly related to biodiversity effects, though releases of different strains and releases of saline versus freshwater would have different effects in different ecosystems.

12.4.4.3 Air Quality

Air quality indicators relate to regional human health, occupational health, or ecosystems. Air emissions can occur during feedstock production, but also during processes such as drying and extraction, refining, and transportation and use. A suite of four indicators has been proposed to measure air quality related to algal biomass production, namely tropospheric ozone, carbon monoxide, total particulate matter (PM) less than 2.5 micrometers (µm) in diameter (PM2.5) and total particulate matter less than 10 µm (PM10) (table 12.1 and Efroymson and Dale 2015). These are the same as the air-quality indicators recommended for terrestrial biomass by McBride et al. (2012). However, even less information is available on these indicators in the context of algal biomass production. The NRC Committee on Sustainable Development of Algal Biofuels (NRC 2012) suggested that air quality indicators may include concentrations of volatile organic compounds (VOCs) and odorous secondary metabolites for open-pond systems. For later steps in the supply chain, the NRC (2012) suggested particulates for active drying processes, air concentrations of solvent used for extraction processes; and particulates, hydrocarbons, and acid gases for pyrolysis, if used, as air-quality-related metrics. The GREET model estimates emissions of six EPA criteria pollutants: CO, VOCs, nitrogen oxides, sulfur oxides, PM10 and PM2.5 (Frank et al. 2011), without a judgment about their relative importance compared to other measures.

However, little evidence exists of emissions of these chemicals and materials from the cultivation process. VOCs have been detected as emissions from open ponds (personal communication from Paul Zimba in NRC 2012).

12.4.4.4 Primary Productivity

Productivity is a measure of the efficiency of biomass or biofuel production, and it may also be an economic or environmental measure (Efroymson and Dale 2015). For photosynthetic organisms, yield of biomass (and ultimately, fuel) is related to primary productivity, i.e., net flux of carbon from the atmosphere to the organisms per unit time. Whether productivity of algae represents an environmental indicator of bioenergy sustainability relates to the extent to which algal biomass cultures are connected to the ecosystem. The ecosystem context of the unit farms in volume 1 of *BT16* is important but outside of the scope of this chapter.

In *BT16* volume 1, the productivities of algae were modeled using the Pacific Northwest National Laboratory's Biomass Assessment Tool, with current productivity and future high-productivity scenarios. Productivity of algae is influenced by abiotic environmental conditions, including temperature and light; biotic conditions such as algae strains; microbial community structure; and the abundance of predators, pathogens, and self-shading by other algae (Kazamia et al. 2012; Shurin et al. 2013). In *BT16* volume 1, higher productivities were observed in warmer, sunnier regions of the United States, and seasonal cycles were projected. Higher productivities are related to higher profitability, and as described above, higher EROI, lower GHG emissions per biomass, and lower water consumption per biomass.

Primary productivity from vegetation on land that was removed to transition to algae production is an important consideration, not only for estimating changes in greenhouse gas emissions (Arita et al. 2016), but also for ecosystem functions. Using lands with high primary productivity could affect higher trophic level animals. Algae cultivation can use land that is marginally productive, reserving highly productive or biodiverse lands for other uses.

12.5 Summary and Future Research

Little information is available to support a quantitative analysis of the environmental effects of algae cultivation. Few examples of commercial algae production exist, and few environmental indicators have been measured for those systems. More specifically, environmental effects of the scenarios from *BT16* volume 1—namely those that involve one of three CO_2 co-location sources (coal-fired power plants, natural gas plants, and ethanol plants), freshwater or saltwater strains (*Chlorella sorokiniana* and *Nannochloropsis salina*), full plastic pond liners or minimal liners, and current or future productivities—have not been measured. GHG emissions from scenarios similar to the *BT16* scenarios have been estimated by Davis et al. (2016), but the full supply chain and appropriate baselines would have to be selected for net emissions to be estimated. Water consumption has been estimated and described for the scenarios in *BT16* volume 1, but the context of those estimates with respect to competitive use has not been determined.

Some conclusions about the scenarios in *BT16* volume 1 are clear. Increasing productivity has benefits for water consumption on a per-unit-biomass basis. Information is available to allow analysts to quantify the difference in GHG emissions between co-location scenarios and carbon-capture scenarios and between supplying pure, captured CO_2 to algae facilities in tanks and piping dilute, impure gas a short distance to algae-cultivation systems. Similarly, information is available to allow the quantification of carbon emissions from plastic liner production. However, quantitative estimates of the GHG emissions of biomass alone are not possible for an algal biomass system that is highly integrated, so a life-cycle analysis would need to evaluate the whole supply chain for CO_2 co-location scenarios. Cultivation systems that use saline or brackish water media have the potential to consume less water than freshwater systems. However, realizing this water consumption benefit would be dependent on the method used to handle blowdown. If evaporation ponds are used, saline water systems will have a relatively high consumptive use.

Some of the indicators require more regional context than others. Water flows are regional and season-specific. Measuring water consumption does not depend on regional variables, but interpreting the environmental significance is a regional exercise. Biodiversity indicators must be selected with the region in mind because particular species and habitats are valued in specific regions. However, GHG emissions have more global significance than regional significance, and understanding the regional context is not important for modeling or measuring emissions from facilities. Similarly, the temporal context of water withdrawals and water consumption is important within a given region. Understanding the relationship between regional soil biogeochemistry and the probability of sealing or leakage of unlined ponds is an important research need tied to water quantity, water quality, GHG emissions and profitability.

As with any environmental assessment, it is important to define a baseline or reference scenario. A business-as-usual scenario was not evaluated in *BT16* volume 1. Therefore, the only qualitative comparisons that can be made here are between the scenarios that were evaluated. The three co-location scenarios in volume 1 do not represent all of the potential algae production co-location strategies. Additional sources of CO_2 may be available (e.g., ammonia plants and cement plants), which would allow more biomass to be generated from potential GHG emissions, and co-location with wastewater is an opportunity to improve water quality of natural waters.

While tradeoffs were not specifically evaluated for the scenarios, it is clear that the cultivation of freshwater algae species uses much more water than the cultivation of saline water species. However, to maintain salinity targets, freshwater may be needed to dilute saline water, or salt may need to be added to brackish water. Much more energy may be spent pumping saline water overland from the sea or from deep saline aquifers to maintain salinity targets than the energy needed to withdraw and transport fresh surface water.

Of course, the exclusion of PBRs has a large effect on the potential national biomass, and enclosed systems have very different environmental advantages and disadvantages than those described here. For example, PBRs often use less water per unit of algal biomass produced, given that they are not subject to appreciable evaporation. However, spray cooling can increase that water use.

The broad spectrum of energy and food production and its intrinsic tie to water and energy use leads to an increasing need to evaluate aspects of sustainability and implement planning strategies. Research needs include quantifying uncertainty in surface and groundwater sources; evaluating, with a spatial and temporal emphasis, the available supplies and non-stationarity climate and extreme events that impact those supplies; identifying existing and proposed food and energy uses; and identifying interactions between uses and sources of water (GAO 2012; Bauer et al. 2014).

In addition to further research on GHG emissions and water quantity (both consumption and natural stream flows), research, including field studies and modeling, is needed to evaluate potential aquatic and terrestrial biodiversity, air quality, water quality, and primary productivity effects of growing diverse species of algae at the commercial scale. A better understanding of environmental effects will allow future resource analyses to quantify the potential availability of more environmentally sustainable biomass, rather than all potential biomass. Such an understanding will help industry place facilities in the best locations and continue to develop good management practices.

Research needs for algae production include quantifying the environmental effects that are only described in qualitative terms in this report and estimating environmental effects in additional contexts to those in the scenarios. As algae-produced food (protein) and feed becomes commercially viable, understanding the interactions between the profitability, food security, energy security, and water quantity will become paramount, just as current research is investigating the water-energy-food nexus.

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Appendix 12-A: Water Resource Indices

Water Resources Vulnerability Index

The Water Resources Vulnerability Index, often referred to as the "withdrawal to availability (WTA) ratio," is probably the simplest and most widely used of the water resources indices. All other indices described here are variants of the basic ratio of water supply to demand (Rijsberman 2006). The Water Resources Vulnerability Index is simply defined as a ratio (equation 12A.1).

Equation 12A.1:

W is total annual water withdrawals, and Q is the sum of available water. In general, a ratio >0.4 indicates water

$$\frac{W}{Q}$$

stress (Raskin et al. 1997; Alcamo et al. 2000). This simple equation can be expanded in a number of ways to include sector-specific water demand (including environmental flows) represented as withdrawals, with a weighting term to indicate estimated fraction of consumptive use and, thus, allowing for a water-reuse term. In addition, the index could be applied to a monthly time scale and at a user-defined spatial scale.

Water Supply Stress Index (WaSSI) and Water Supply Stress Index Ratio (WaSSIR)

The Water Supply Stress Index (WaSSI), originally proposed by Sun et al. (2008a, 2008b) provides a measure of the relative supply and demand of water at a monthly time step for eight-digit Hydrologic Unit Code (HUC) watersheds. The WaSSI is defined as equation 12A.2.

Equation 12A.2:

$$WaSSI_x = \frac{WD_x}{WS_x}$$

WD is water demand, *WS* is water supply, and *x* represents any number of different simulations that might impact water availability. In the original context of the equation, *x* identified baseline conditions and simulations around future changes to climate, land use, population, and various combinations of these changes. Traditionally, the use of this index is hinged on the use of the sector-specific U.S. Geological Survey (USGS) 5-year water-use data (e.g., Maupin et al. 2014), though other water-demand data could be used if they are available. *WD* then is defined by equation 12A.3.

Equation 12A.3:

$$WD = \left(\sum_{i=1}^{n} WU_x\right) + \sum PB_x$$

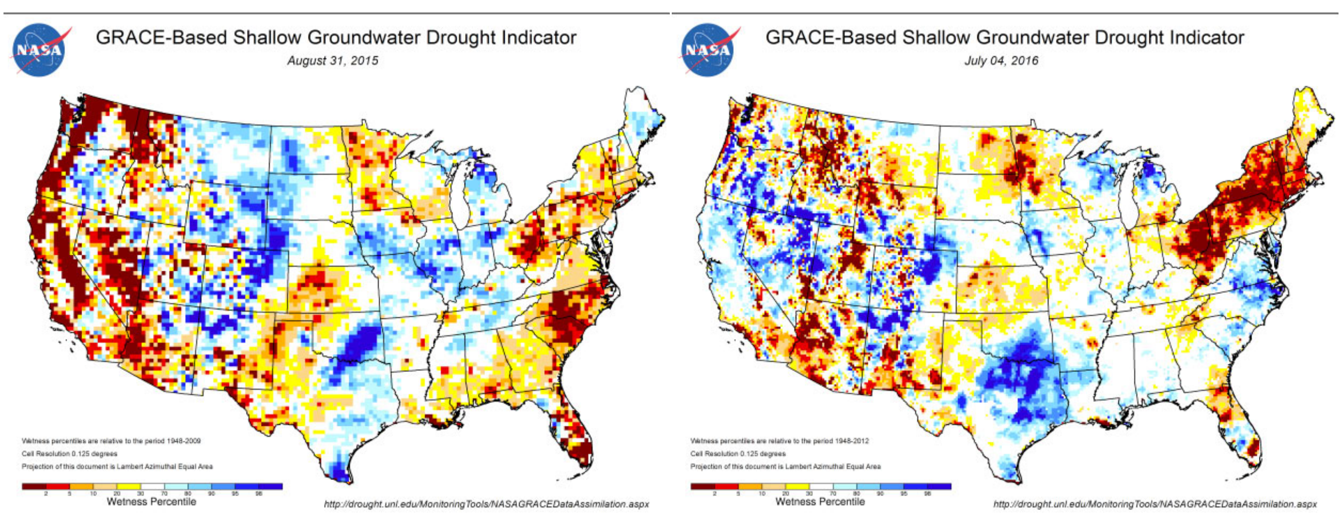
WU is the summation of water use by USGS-defined water-use sector n for each simulation, *x*, and *PB* is public use that also considers inter-basin water transfers by evaluating the difference between public-use withdrawals and public water supply. Water supply (*WS*) is defined by equation 12A.4.

Equation 12A.4:

$$WS = SS + GS + \sum RF_x$$

For a given watershed boundary, *SS* is the surface-water supply volume (e.g., eight-digit HUC), *GS* is the groundwater supply, and *RF* is the return flow volume for each sector *x*. The surface-water supply volume is a measure of total precipitation in the basin less the evapotranspiration lost out of the basin. Evapotranspiration can be calculated in one of a number of ways: Penman-Monteith, mass/energy balance models, models using observed satellite data (Moderate Resolution Imaging Spectroradiometer [MODIS] – MOD16 Global Evapotranspiration; Surface Energy Balance Algorithm for Land [SEBAL]; Mapping EvapoTranspiration at high Resolution with Internalized Calibration [METRIC]). The groundwater supply can be estimated in many ways (i.e., models, direct and remote-sensed observations [fig. 12A.1]); however, for sustainable use, Alley, Reilly, and Franke (1999) define this as "development and use of groundwater in a manner that can be maintained for an indefinite time without causing unacceptable environmental, economic, or social consequences," requiring more local-to-regional definition. Sun et al. (2008a) and McNulty et al. (2010) suggest use of withdrawal records from the USGS 5-year water-use data (Solley et al. 1998; Maupin et al. 2014).

Figure 12A.1. | Regular monitoring of shallow groundwater supply using a data assimilation of the National Aeronautics and Space Administration's GRACE (Gravity Recovery and Climate Experiment) satellite data, direct observations, and mass/energy numerical models (National Drought Mitigation Center, University of Nebraska – Lincoln; Houborg et al. 2012)



Finally, the return flows can be estimated as a fraction of the sector-specific water use, and it is suggested that this fraction be established regionally per sector.

To complement the WaSSI, the Water Supply Stress Index Ratio (WaSSIR) provides a ratio between a baseline and the various simulations of the WaSSI. The WaSSIR is simply defined as equation 12A.5.

Equation 12A.5:

$$WaSSIR_x = \frac{WaSSI_x - WaSSI_b}{WaSSI_b}$$

Where $WaSSI_x$ is a given WaSSI scenario and $WaSSI_b$ is the baseline, negative ratio values indicate reduced water stress, and positive values indicate increased water stress as compared to baseline. The further positive deviation from 0 there is, the greater the stress.

Potential simulations using the WaSSI can not only incorporate future projected climate, population, and land-use chance (LUC), but can also specifically address and evaluate the potential water sustainability of potential microalgae production from scenarios in the *2016 Billion-Ton Report* (*BT16*) volume 1. The site-specific, modeled, consumptive water-use requirements (Wigmosta et al. 2011) for each carbon dioxide (CO_2) co-located site can be aggregated from the site scale to the watershed scale, and the WaSSIR can be assessed to determine the potential water stress.

Water Scarcity Index (Wsci)

Asheesh (2007) established the Water Scarcity Index (Wsci) as a method to identify gaps that would prevent a balance in water supply and water demand amongst a complex relationship of variables, referred to as the Water Equality Accounting System. The Wsci is defined by equation 12A.6.

Equation 12A.6:

$$Wsci = \left(\frac{\alpha}{\left(\left[\frac{100}{100-p}\right] \beta e^{\lambda \Delta t} (\varepsilon + \gamma + \delta) \left(\frac{100}{100-\kappa} \right) + h + b \right)} \right) - 1$$

Where the following are true:

α is the annual total freshwater availability

p is the industrial water demand (%)

β is the current population

λ is the population growth rate

Δt is the length of time considered (years)

k is the estimated annual water system losses

ε is the annual domestic per capita demand

γ is the annual per capita demand for green areas (dependent on population growth)

δ is the annual irrigation demand

h is the annual evapotranspiration

b is the annual environmental water requirement.

While there is no specific spatial scale identified in the Wsci equation, this could be evaluated on a watershed-to-regional scale where information is available to support the terms. In addition, the equation was intended to be run at an annual scale with a future projection of population, and it would be possible to evaluate the use of the Wsci at the monthly time-step to better reflect water opportunities for microalgae production. Modeled, net consumptive water use of open pond or photobioreactor (PBR) systems can be incorporated into a new, independent term for the denominator of this equation.

Water Stress Indicator

Smakhtin et al. (2005) provide a simple environmental water-scarcity method forward that considers the relationship of water withdrawals to the environmental water requirement: the Water Stress Indicator (WSI). The WSI is represented by equation 12A.7.

Equation 12A.7:

$$WSI = \frac{W}{MAR - EWR}$$

Where *W* is the total water withdrawal in a basin, *MAR* is the naturalized, long-term mean annual runoff volume that represents the total water supply, and *EWR* is the annual environmental water requirement. The WSI method is intended for a global-scale analysis, but as with other indices included in this chapter, it can be modified for use at finer temporal or spatial scales to help understand the water-use impacts that microalgae production might have in the spatiotemporal context. The environmental water requirements can be defined in a number of ways, as described below (environmental flow requirements). The classification of the WSI is described in table 12A.1.

Table 12A.1. | Classification of the Water Stress Indicator (WSI) as Defined by Smakhtin et al. (2005)

WSI	State of Basin Environmental Water Scarcity
WSI > 1	Overexploited (current water use is tapping into EWR)— environmentally water-scarce basins
0.6 ≤ WSI < 1	Heavily exploited (0%–40% of the utilizable water is still available in a basin before EWRs are in conflict with other uses)—environmentally water-stressed basins
0.3 ≤ WSI < 0.6	Moderately exploited (40%–70% of the utilizable water is still available in a basin before EWR are in conflict with other uses)
WSI < 0.3	Slightly exploited

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Appendix 12-B: Environmental Flow Requirements

As described in chapter 7, environmental flow requirements are a key target for environmental indicators. The International Union for Conservation of Nature and Natural Resources defines environmental flows as "…the water regime provided within a river, wetland, or coastal zone to maintain ecosystems and their benefits where there are competing water uses and where flows are regulated. Environmental flows provide critical contributions to river health, economic development, and poverty alleviation. They ensure the continued availability of the many benefits that healthy river and groundwater systems bring to society" (Dyson et al. 2003). Thus, indicators of environmental flow may be considered indicators of biodiversity, as well as indicators of water quantity, and need to be considered in sustainable use of water resources. In addition, environmental water requirements identified in several of the water-resource indices previously described can be assessed using a variety of methods noted below.

A fundamental tenet in developing environmental flow requirements is the understanding that hydrologic conditions have intrinsic variability over a range of time scales, primarily as a function of meteorology and longer-term climate. The management of environmental flows involves understanding the components of flow and their relationship to the specific ecosystem need—namely magnitude, frequency, duration, variability, timing, and rate of change—which can impact the overall structure and function within an ecosystem (Poff and Ward 1989; Richter et al. 1996). More specifically, fluxes of nutrient and prey availability; habitat development and maintenance; life-history flow requirements that support migration freshets, spawning, and nursery environments; flushing flows for sediment cleaning, transport, and redistribution; hydrologically connected/fragmented habitats; water quality; quality of aquatic habitats; riparian and wetland function; and more can contribute to the environmental flow requirements (O'Keeffe and Quesne 2009).

Peak flow and minimum base flow are described as basic environmental indicators for water quantity in table 12.1, but more complex measures may be needed to incorporate the regional context. Tharme (2003) identified more than 200 methods available to assess environmental flows, and generally, they can be classified as hydrological, hydraulic rating, habitat simulation, and holistic methodologies. The methods can be distinguished as follows:

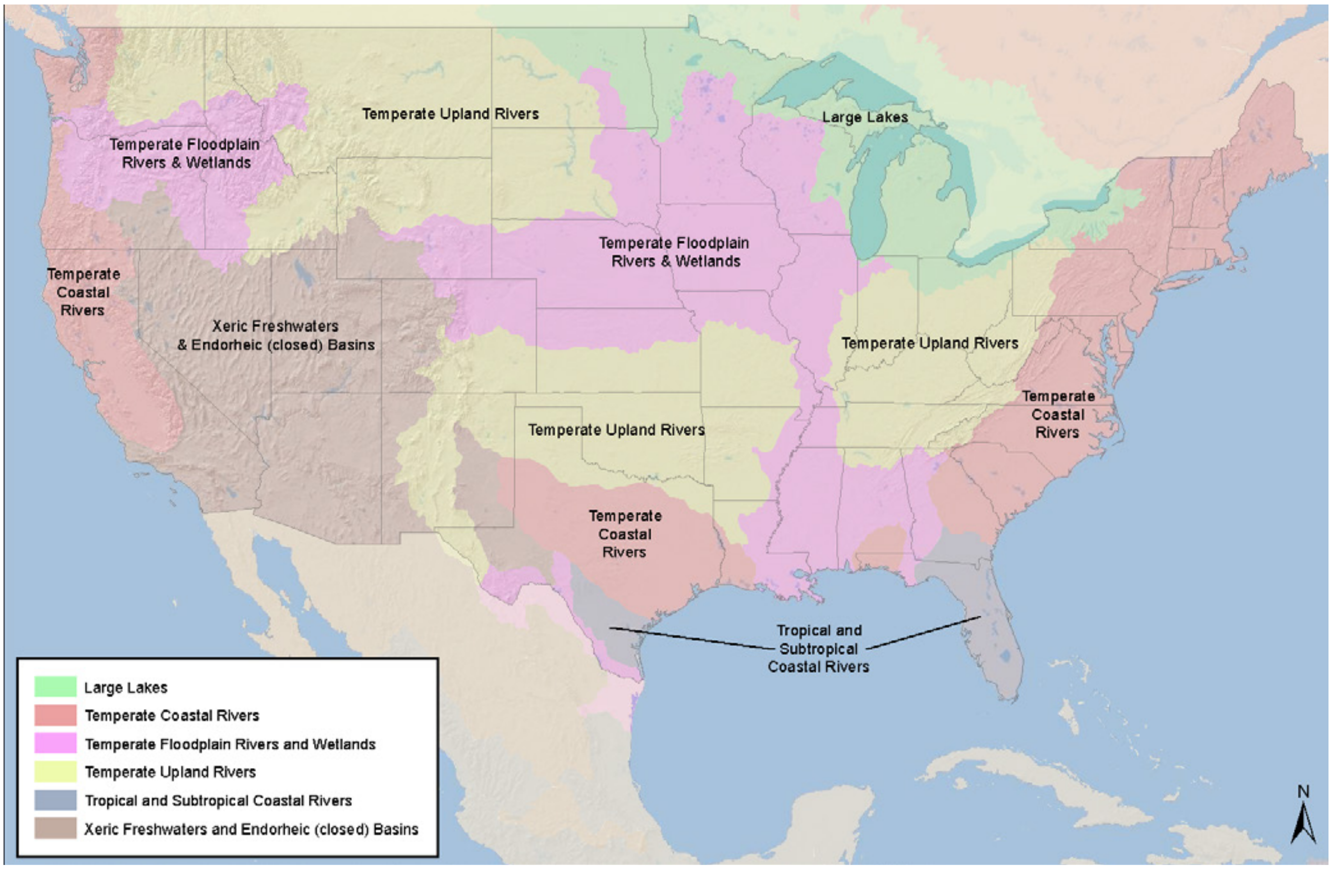
- **Hydrological:** Typically, the hydrological methods are the simplest within the four groups of environmental flow–assessment methodologies. These methods primarily use observed or modeled flow data as in the form of naturalized (unimpaired) flow, historical daily or monthly flow records. Some methods also incorporate a combination of catchment-scale variables and geomorphic and biological indices. In general, these methods are appropriate for planning-level applications and can use readily available data. A few examples of these models follow: (1) The Q_{95} Index environmental flow is defined as the flow that exists ≥95% of the time during the period of record. (2) The Tennant Method (Tennant 1976) assesses environmental flow as a percentage of mean annual flow that relates to the desired habitat quality, where 10% is poor quality, 30% is moderate, and >60% is excellent; however, these values were established statistically for rivers and streams in the midwestern United States and would need to be adjusted for other regions. (3) The Indicators of Hydrologic Alteration (Richter et al. 1996) use numerous input parameters and more than 60 resulting indicators to provide a statistical characterization of the system of interest.

- **Hydraulic Rating:** This class of methods relies on more-detailed and site-specific information, requiring channel cross-sections or three-dimensional bathymetry to assess and relate changes in simple hydraulic variables (e.g., wetted perimeter) to define habitat or aquatic-system requirements and objectives. Environmental flows then are established by defining habitat quality thresholds by relating discharge to habitat indices of concern. The flows generally follow a response curve, or flows are set according to a fixed percentage to reflect an acceptable loss of habitat and/or habitat function. Much of the work done under the hydraulic rating methods is preparatory for more complex method groups of habitat simulation and holistic approaches. The R2Cross method is an example of a hydraulic-rating method (Espegren 1996; Armstrong, Todd, and Parker 2001).
- **Habitat Simulation:** Detailed, local-scale analyses that incorporate hydrological data (i.e., flow magnitude, frequency, duration, variability, timing, and rate of change), hydraulic data (i.e., depth, velocity, shear stress, etc.), and biological data (i.e., habitat-suitability index for specific species, assemblage of species, life stages) are used together to produce habitat-discharge curves and habitat-exceedance probability curves. Well-known simulation models here include the Instream Flow Incremental Methodology (IFIM) model/Physical Habitat Simulation (PHABSIM) model (Bovee et al. 1998; Milhous and Waddle 2012) and variants, including the River Hydraulics and Habitat Simulation (RHYHABSIM) model (Jowett 1989), the Riverine Habitat Simulation (RHABSIM) model (Payne 1994), and the Mesohabitat simulation (MesoHABSIM) model (Parasiewicz 2001, 2007). This approach brings together a more detailed emphasis on flow thresholds that best support system biodiversity. Generally, because of the data and effort required, an approach such as this is conducted at a more local scale.
- **Holistic:** The goal of holistic methodologies, such as the building block method (King, Tharme, and de Villiers 2000, 2008), is to consider the various aspects—ecological, geomorphological, and social—of an entire riverine ecosystem in order to develop appropriate levels of environmental flow. These are often constructed across a range of possible conditions that require expert judgment from multiple subject-matter experts and may ultimately move toward an adaptive management plan. Two general approaches are taken: (1) top-down, where environmental flows across a range of different conditions are defined as an acceptable departure against naturalized or reference flows, as categorized by critical flow events from the perspective of multiple subject matter experts; and (2) bottom-up, where defined objectives of individual elements (ecological, geomorphic, socioeconomic, etc.) are built up and assessed at a finer temporal resolution (i.e., monthly or seasonally) in order to achieve a multi-objective flow regime. As with the habitat simulation approaches, the holistic methodologies are time- and resource-intensive and are not effectively used at a regional or national scale.

Because each aquatic system has specific requirements and objectives, deriving physical and ecological interrelationships requires expert opinion and/or field-collected data to fully parameterize more local-scale hydraulic and habitat models. It also requires appropriate regional- to national-scale analyses of environmental flow requirements, often using a "desktop method" (also known as "historic flow," "discharge," and "lookup table" methods) that only requires the use of consistent and readily available data, such as that found in the national sets of hydrologic data from the USGS's National Water Information System (http://waterdata.usgs.gov/nwis). It is possible, however, to incorporate an ecological component into some hydrologic-based environmental flow methods via weighting factors or percent water allocations. A biogeographical classification dataset such as the

Freshwater Ecoregions of the World (Abell et al. 2008) provides a regional perspective that incorporates aquatic biodiversity, endemism, and hydrologic fragmentation (see fig. 12B.1), thus allowing varying ecological functions and conditions to be evaluated separately.

Figure 12B.1. | A global freshwater ecoregion classification that incorporates aquatic biodiversity, endemism, and hydrologic fragmentation (Source Data: Abell et al. 2008)



Pastor et al. (2014) evaluated several hydrologically based environmental flow–assessment methods, including Smakhtin, Revenga, and Döll (2004), Tennant (1976), Hoekstra et al. (2012), Tessmann (1980), and their own variable monthly flow (VMF) and Q90_Q50 (flow that is present 90% of the time; flow that is present 50% of the time) methods. The Tessmann (1980) and VMF approaches are the only methods that consider monthly flow, low-flow, high-flow, and intermediate-flow indicators. These two methods are based on a temporal resolution that reflects inter-annual cycles; they represent a more detailed flow regime, fit the requirement of using readily available data for basin-, regional-, or national-scale analyses, and have been demonstrated with a high correlation to locally developed environmental flows amongst several characteristically varying basins. We discuss these two approaches below.

Variable Monthly Flow (VMF) Method

The VMF method is one suggested approach for estimating environmental flow. It uses the variability in monthly flow to quantify a reasonable level of ecological protection with the ability to support other water-use activities, such as agriculture, industry, and domestic use. The VMF provides temporal detail (monthly) that is appropriate for planning around the flows throughout the year and allocates required environmental flows as a percentage of the naturalized mean monthly flow. When combined with estimates of existing consumptive water use across all sectors, these estimates provide a high-level assessment of available freshwater resources that is appropriate for use at the regional and national scales for the purpose of resource use and planning; however, particular local conditions and policies will always need to be evaluated. To reflect the variability and ecological need in seasonal flows, in general, a smaller percentage of flows can be allocated for socioeconomic use during low-flow periods; during high-flow periods, a greater fraction of the water resource can be used. Allocation rules can be adjusted according to regional conditions and need across both environment and socioeconomics; general guidelines are provided in table 12B.1.

Table 12B.1. | General Flow Allocation Guidelines for the VMF Method Assessed Monthly (Pastor et al. 2014).

	% of Mean Annual Flow	% Water for Environmental Flow	% Water for Socioeconomic Use
Low-Flow	<40%	60%	40%
Intermediate-Flow	40%–80%	30%–60%	40%–70%
High-Flow	>80%	30%	70%

Tessmann Method

The Tessmann (1980) method to assess environmental flow is a modification of an earlier method established by Tennant (1976), which is based on the field assessment of nearly a dozen rivers in Montana, Nebraska, and Wyoming. A fraction of the mean annual flow is required, where, generally, 10% is the minimum flow and intended only for short-term use to sustain the aquatic environment, and ≥30% of the mean annual flow is what is required to sustain the ecological integrity of the aquatic ecosystem. Additional guidance was provided by Tennant (1976) for low-flow and high-flow seasons, fall and spring, respectively, to maintain the aquatic ecosystem (see table 12B.2).

Table 12B.2. | Tennant (1976) Recommendations for Environmental Flows Parsed by Low- and High-Flow Seasons

Aquatic Ecosystem Maintenance	% of Mean Annual Flow	
	Low-Flow (Oct.–Mar.)	High-Flow (Apr.–Sep.)
Flushing Flows, or Maximum	200%	200%
Optimum Range	60%–100%	60%–100%
Outstanding	40%	60%
Excellent	30%	50%
Good	20%	40%
Fair/Degrading	10%	30%
Poor/Minimum	10%	10%
Severe Degradation	<10%	<10%

Tessmann (1980) uses the same principles as Tennant (1976), but instead of using two flow regimes (low-flow, high-flow) and mean annual flow, Tessmann (1980) uses a ratio of mean monthly flows to mean annual flows and, accordingly, assigns flow rules to one of three categories (see ruleset in table 12B.3). With more temporal detail, Tessmann (1980) offers an approach that can be used in a variety of hydrologic systems throughout the world, though in general, the environmental flow guidelines are more conservative than other methods discussed here (i.e., it keeps more flow in the river).

Table 12B.3. | The Tessmann (1980) Rules for Environmental Flow Based on Naturalized Mean Monthly Flow (MMF) and Naturalized Mean Annual Flows (MAF)

Naturalized Flow Condition	Environmental Flow Requirement
if: MMF < 40% MAF	*then*: MMF
if: MMF > 40% MAF and 40% MMF < 40% MAF	*then:* 40% of MAF
if: 40% MMF > 40% MAF	*then:* 40% of MMF

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Appendix 12-C: Contributions of Sectors to Total Consumptive Water Use

Figure 12C.1 illustrates the percentage of each consumptive water-use category contributing to the total consumptive water use from 1985 to 2000. All years except 2000 contain consumptive use data; therefore, a trend for each category was determined as a ratio of consumptive water use to total water use and extrapolated to determine the sector-specific consumptive water use for 2000 (Moore et al. 2015). Comparing consumptive water use to water withdrawal provided in the U.S. Geological Survey (USGS) 2010 water-use report (Maupin et al. 2014), one can see the significant fractional difference in consumptive use vs. withdrawals, where, for example, thermoelectric represents 3.9% of all consumptive use; for withdrawals, this same sector accounts for ~45% of all withdrawals (i.e., a significant amount of water is pulled for thermoelectric use, but is returned to the system) (fig. 12C.2).

Figure 12C.1. | Percentage of average annual consumptive water use for each USGS-defined sector, 1985–2000

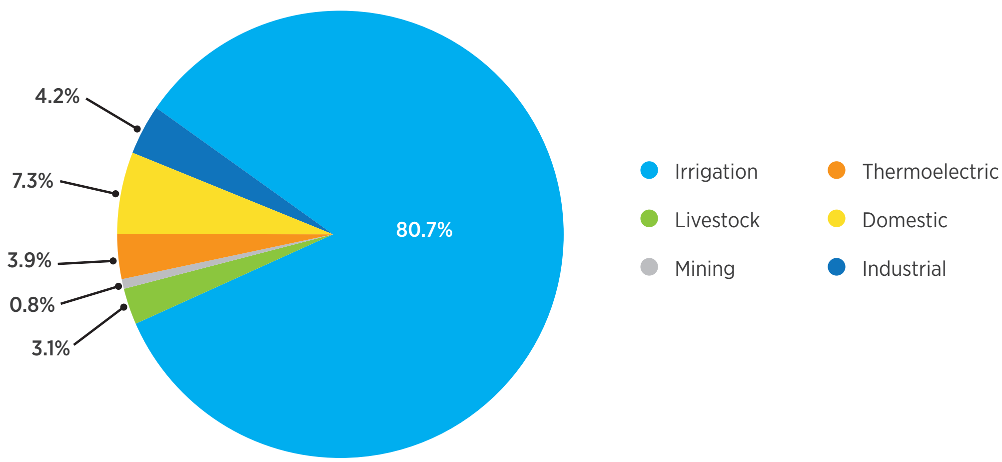
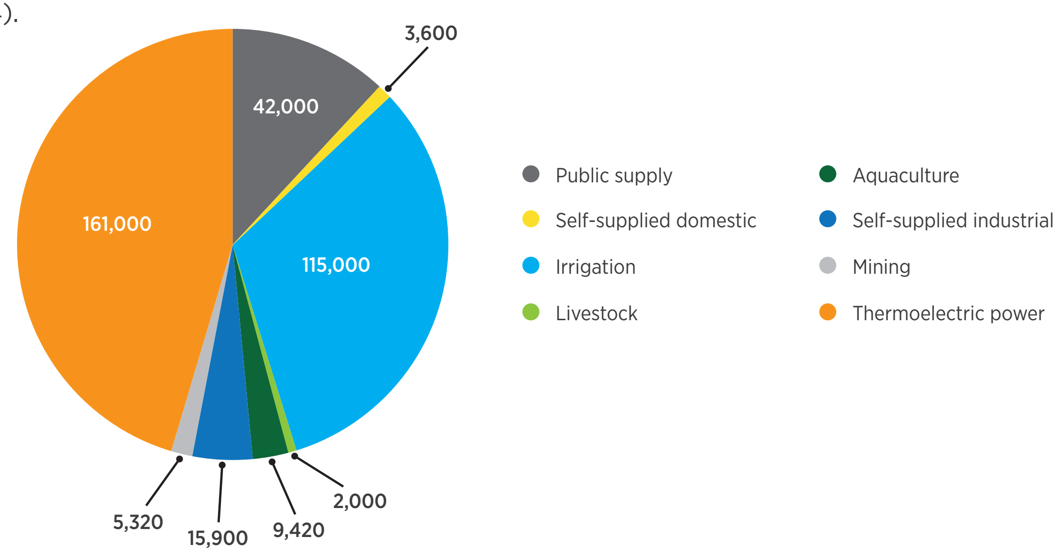
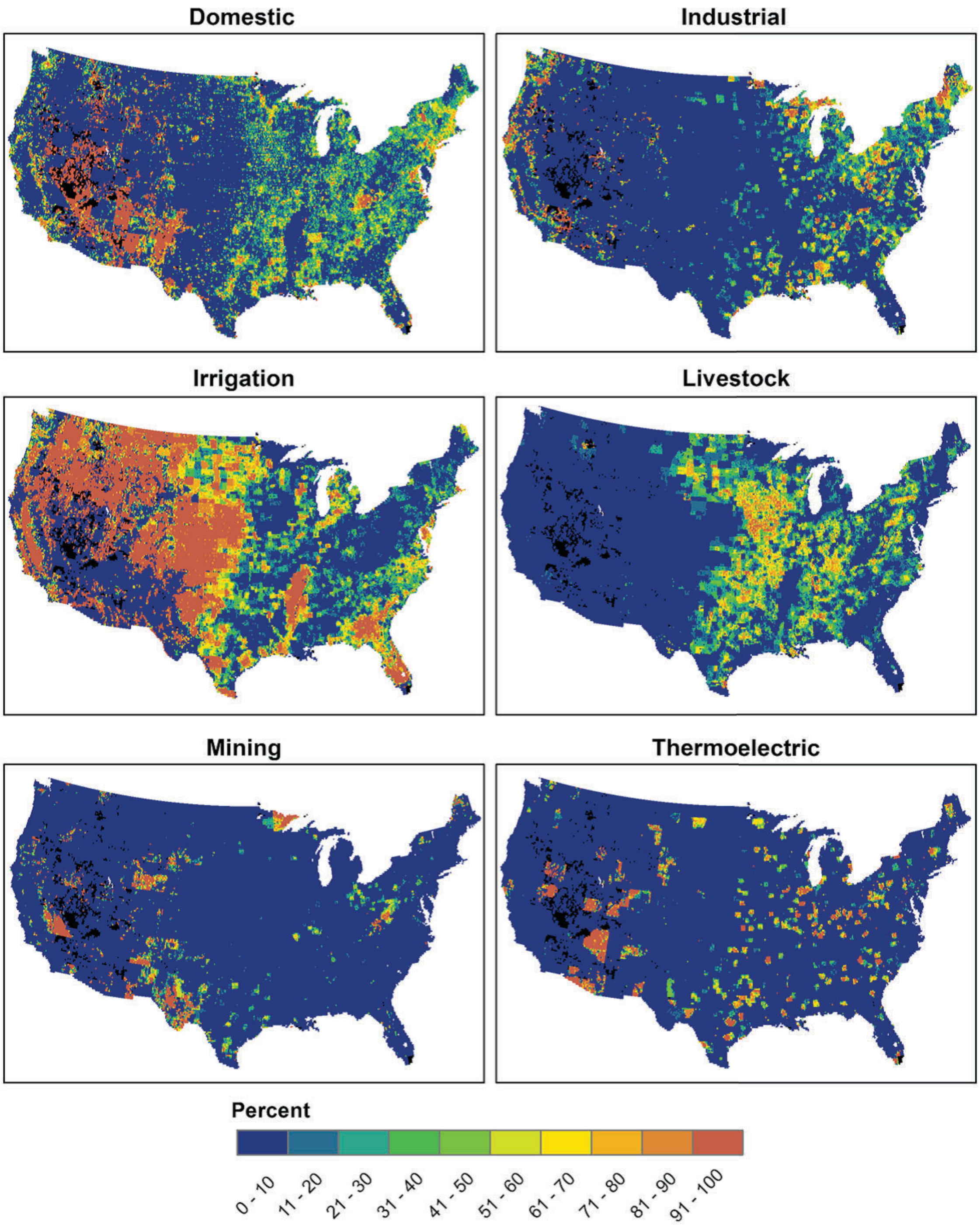


Figure 12C.2. | The pie graph represents the USGS 2010 estimates of water withdrawals per sector averages, 2005–2010. Note the additional sectors in the 2010 data as compared to the 1985–2000 data. (Image credit: Maupin et al. 2014).



To help understand the distribution of consumptive water use amongst sectors, Moore et al. (2015) developed a spatially explicit view of the percent annual consumptive use relative to the total as an average from 1981 to 2000 (fig. 12C.3).

Figure 12C.3. | A spatially explicit high-resolution (1/8°) sector view showing percentage of annual consumptive use relative to the total consumptive water use. Values represent an average from 1981–2000 and highlight primary consumptive water-use sectors within the United States. (Image credit: Moore et al. 2015).



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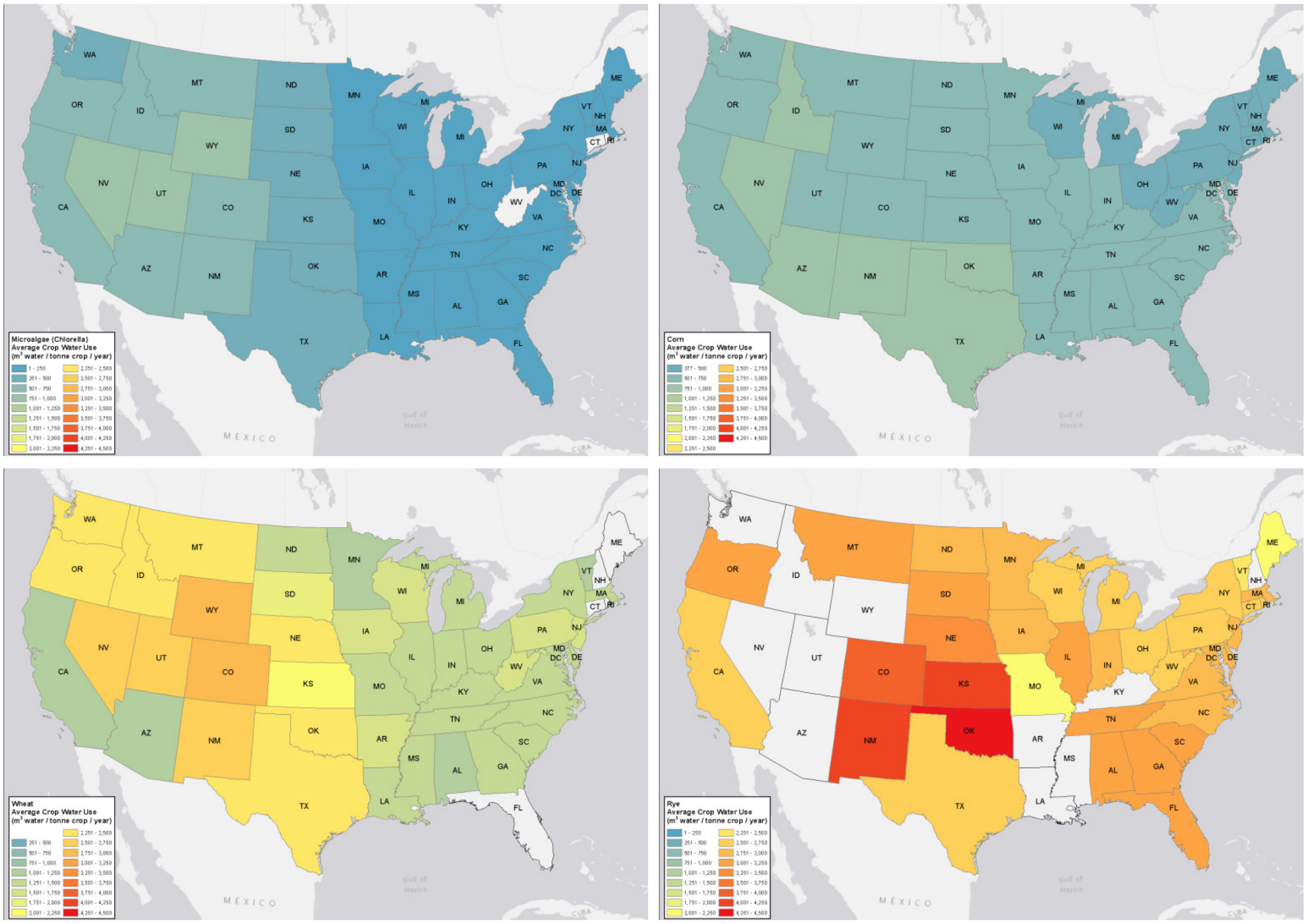
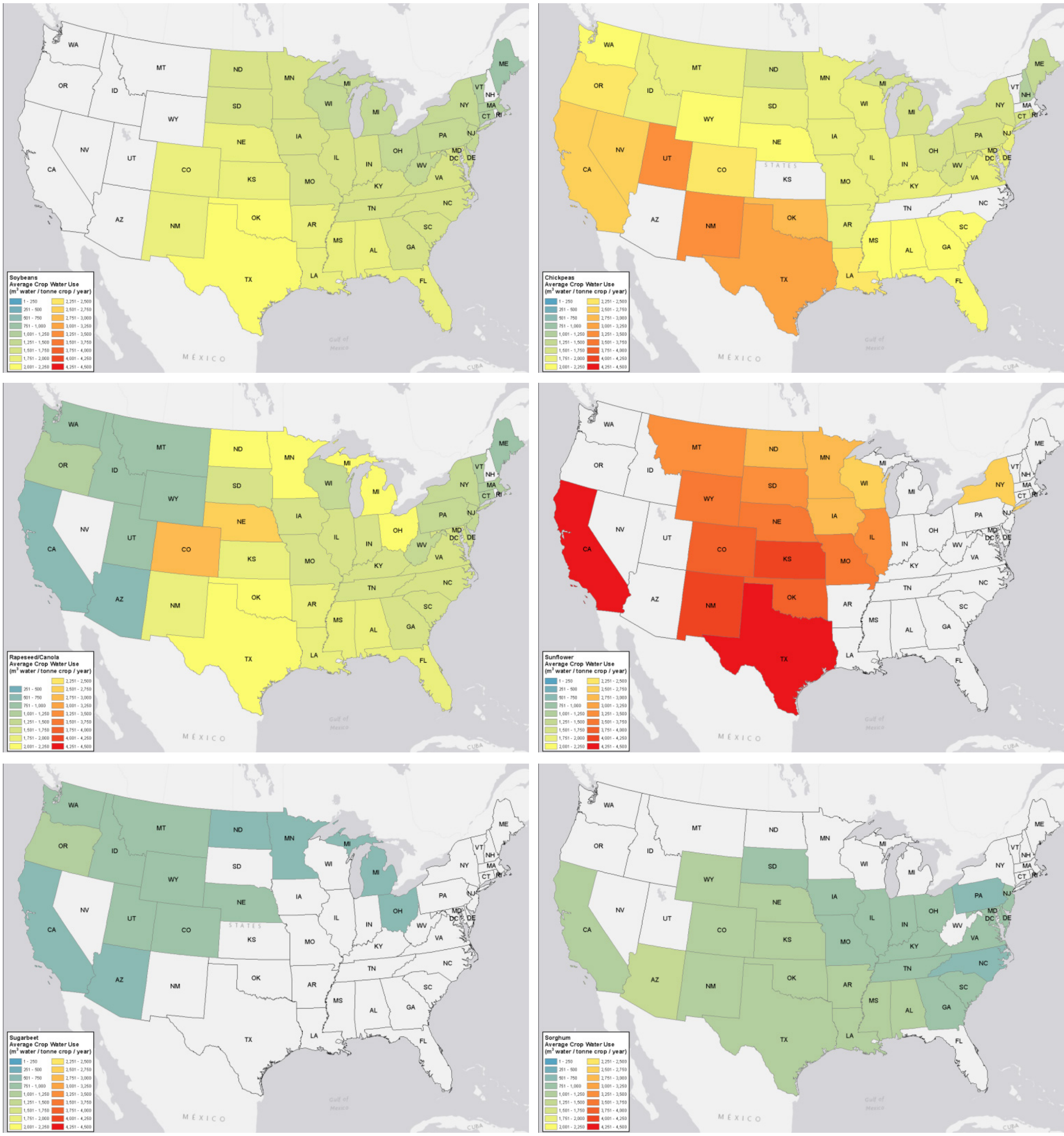
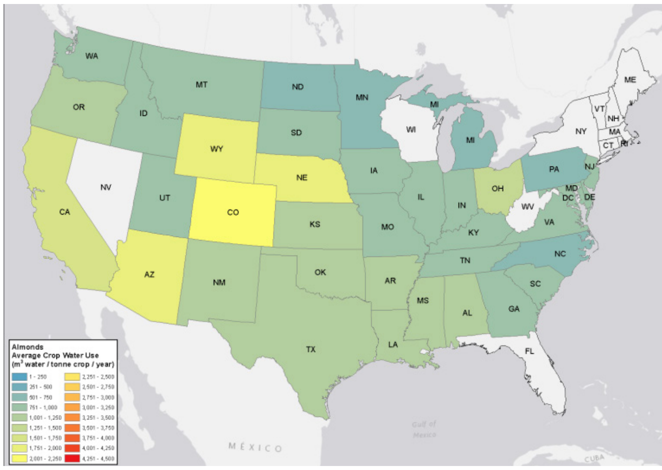
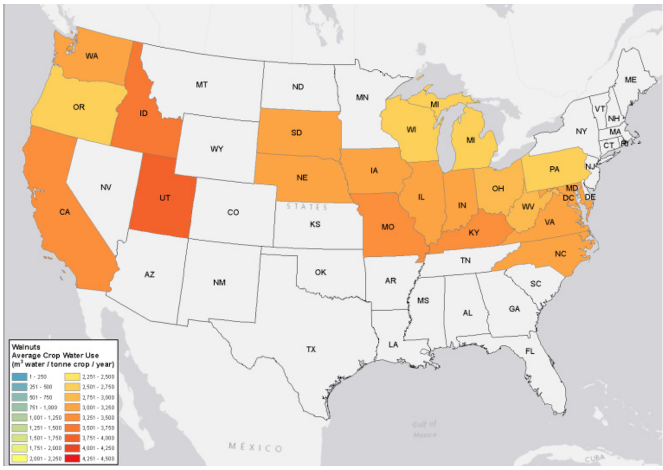
Appendix 12-D: Comparison of Water Use by Selected Terrestrial Crops and Microalgae Water Use—Geographic Analysis

Figure 12D.1. | Annual average green + blue crop water use by state. Terrestrial crop water use from Mekonnen and Hoekstra (2011); microalgae crop water use from Wigmosta et al. (2011).




Soybeans
Average Crop Water Use
(m³ water / tonne crop / year)
Chickpeas
Average Crop Water Use
(m³ water / tonne crop / year)
Rapeseed/Canola
Average Crop Water Use
(m³ water / tonne crop / year)
Sunflower
Average Crop Water Use
(m³ water / tonne crop / year)
Sugarbeet
Average Crop Water Use
(m³ water / tonne crop / year)
Sorghum
Average Crop Water Use
(m³ water / tonne crop / year)





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