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Bioenergy Development Policy and Practice Must Recognize Potential Hydrologic Impacts: Lessons from the Americas

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Abstract Large-scale bioenergy production will affect the hydrologic cycle in multiple ways, including changes in canopy interception, evapotranspiration, infiltration, and the quantity and quality of surface runoff and groundwater recharge. As such, the water footprints of bioenergy sources vary significantly by type of feedstock, soil characteristics, cultivation practices, and hydro-climatic regime. Furthermore, water management implications of bioenergy production depend on existing land use, relative water availability, and competing water uses at a watershed scale. This paper reviews previous research on the water resource impacts of bioenergy production—from plot-scale hydrologic and nutrient cycling impacts to watershed and regional scale hydro-economic systems relationships. Primary gaps in knowledge that hinder policy development for integrated management of water–bioenergy systems are highlighted. Four case studies in the Americas are analyzed to illustrate relevant spatial and temporal scales for impact assessment, along with unique aspects of biofuel production compared to other agroforestry systems, such as energy-related conflicts and tradeoffs. Based on the case studies, the potential benefits of integrated resource management are assessed, as is the need for further case-specific research.

Keywords Second-generation biofuels · Eco-hydrology · Water quality · Integrated assessment · Hydro-economic modeling

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Introduction

Human activities associated with bioenergy systems affect water availability and water quality from the plot to the watershed scale, leading to impacts on socio-economic and aquatic ecosystems. In turn, the local availability of high-quality water resources can affect the production levels of bioenergy feedstock. The analysis and modeling of bioenergy systems for decision support and policy-making require holistic approaches that consider both human and natural system components (Ng et al. 2010).

Bioenergy production is occurring globally, but it is growing especially quickly in the Americas, due to a variety of policy initiatives. An assessment of land availability for second-generation bioenergy feedstocks—cellulosic and native perennial energy crops—found that

in South America and the U.S., there are 150–220 million hectares of abandoned or degraded agricultural land that could be used for biofuel production, potentially contributing 5–15 % of global liquid fuel consumption (Cai et al. 2011). A variety of bioenergy systems (i.e., crops and production processes) are in place, or being planned, across the region. Brazil is the world's second largest producer of ethanol, primarily from sugarcane, and it has increased production of biodiesel, mainly from soybeans, nearly tenfold since 2007 (Barros 2013). In Argentina, biodiesel is produced primarily from soybeans, but sustainability assessments of large-scale bioenergy production from both soybeans and switchgrass (for pellets) have been completed (Van Dam et al. 2009). In Mexico, jatropha is being evaluated on a pilot scale to use as a biofuel feedstock (Skutsch et al. 2011), and several second-generation conversion technologies have been identified for biofuel production from municipal solid waste (Romero-Hernandez et al. 2012). In the United States and Canada, the dominant biofuel is corn ethanol. However, with advances in cellulosic ethanol and biodiesel technologies, there is potential for woody feedstocks and perennial grasses representing second-generation biofuel crops to contribute significantly to the energy supply (Carriquiry et al. 2011).

Different feedstock types, crop management practices, and biofuel production processes can have highly variable and complex effects on water resources. Some second-generation biofuel crops are known to present tradeoffs in terms of water availability and water quality. For example, miscanthus sequesters nutrients in its root system in winter, thus, resulting in lower nutrient runoff and improvements in water quality relative to alternative cropping systems (Ng et al. 2010). However, miscanthus consumes more water during the growing season than corn, soybean, or other cellulosic crops, resulting in lower runoff and potentially more frequent critical low-flow events (Le et al. 2011). For woody feedstock crops with long rotation cycles (e.g., 10–20 years), water quantity and quality impacts are expected to vary significantly on inter-annual time scales. At the same time, it is well known that some impacts, such as evapotranspiration effects on surface and groundwater systems, also fluctuate over sub-annual time scales. Local hydro-climatic conditions will also play a large role in these time-varying impacts.

Understanding and managing water resources impacts of bioenergy systems at the watershed scale require the use of watershed models, calibrated with measurements of plant water use and groundwater and surface water conditions. Further, integrated tools such as hydro-economic optimization models, though more difficult to calibrate, can aid in the management of socio-economic impacts

(e.g., Moraes et al. 2010). Hydro-economic models have typically been applied at the watershed scale, but recently some features of this approach (e.g., water availability constraints) have been applied in regional, national, and global scale models to guide policy development and renewable energy targets (Nunez et al. 2013; Carneiro et al. 2014).

In evaluating the potential for integrated management of bioenergy-water resource systems, this paper reviews the state of knowledge of water resources impacts of bioenergy development and opportunities for more effective feedstock production and energy conversion strategies (Fig. 1). Because many estimates of water needs for bioenergy feedstock cultivation have been generalized over large scales, we recommend appropriate spatial and temporal scales for impact evaluation and planning. Then, several case studies are reviewed to illustrate opportunities for improved water–bioenergy management through integrated approaches, including economy-wide models at regional, national, and global scales. Based on case study findings, policies for promoting more holistic management are briefly discussed, and a research agenda is outlined to better inform bioenergy policies and decision making.
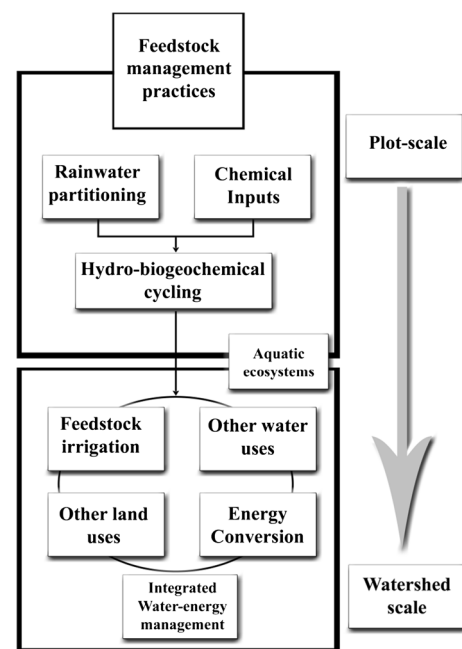
Fig. 1 Integrated management of water–bioenergy systems calls for understanding of water quantity and water quality impacts at a range of scales in order to develop economical and environmentally sound feedstock production and energy conversion processes

Hydrologic Impacts of Bioenergy Systems

Impacts of Feedstock Cultivation on Field Scale Water Budget

Approximately 80–90 % of the evapotranspiration (ET) from the Earth's terrestrial surface is returned to the atmosphere via plant transpiration (Jasechko et al. 2013); consequently, changes in vegetation cover, such as fast-growing woody bioenergy crops replacing annual crops, pasture, or native vegetation, can strongly impact water yield. Impacts of bioenergy crop production on the hydrologic cycle via changes to field level water balance can be quantified using a water budget approach, as follows:

$$\mathrm{WY} = P - T - I - E + \Delta S,$$

where WY is the water yield, P is the precipitation, T is the plant transpiration, I is the rainfall interception by the canopy, E is the soil evaporation, and ΔS is the change in soil water (Fig. 2). Not only water yield is primarily runoff, but it may also include interflow and baseflow, which reach streams after some lag time. Rainfall interception loss is the portion of precipitation that does not reach the ground during and after a precipitation event, and throughfall and stemflow constitute the portion that reaches the ground. Typically, the change in soil water is considered negligible over longer time periods, such as a year.

Several studies have assessed the hydrologic impacts of replacing annual agricultural crops with perennial grass-based bioenergy crops. For example, research in central Illinois, USA estimated that ET from miscanthus is about 120 mm higher than from rotational corn–soybean row-crop systems typical of the region. In contrast, ET from switchgrass is similar to that of corn–soybean systems (about 450 mm) (Dominguez-Faus et al. 2009), but features a seasonal shift, with higher ET early in the spring and lower ET in early summer for switchgrass compared to corn or soybeans (McIsaac and David 2010). As another example, in Sao Paulo, Brazil, sugarcane has been planted on pastures and cropland, and this land-use change causes water availability concerns (Boddey et al. 2008; Rudorff et al. 2010; Egeskog et al. 2014). Although ET is generally lower in cropland compared to native grassland, shrub, or forest vegetation (Smeets et al. 2008; Berndes 2008), sugarcane has a longer growing period, requires a larger amount solar radiation for the sugar conversion process, and therefore has a higher ET than most other crops (Wiedenfeld 2004).

Less emphasis has been placed on assessing the hydrologic impacts of land-use conversion to forest plantation bioenergy systems. The impact of forest plantation-based bioenergy crops on local hydrology depends on the ecosystem being replaced (Jackson et al. 2005; Pielke et al. 2007). In general, short rotation, highly productive forest plantations will result in increased transpiration and reduced streamflow relative to the reference vegetation (Jackson et al. 2005). A shift toward greater carbon sequestration typically results in increased evapotranspiration (e.g., Scott and Prinsloo 2008; Jackson et al. 2005). However, the impact will depend on the difference between the evapotranspiration from the previous land cover and bioenergy crop (Pielke et al. 2007). The closer the bioenergy crop mimics the previous land use, the lesser an

Fig. 2 Components of the hydrologic cycle and water budget of a forest environment
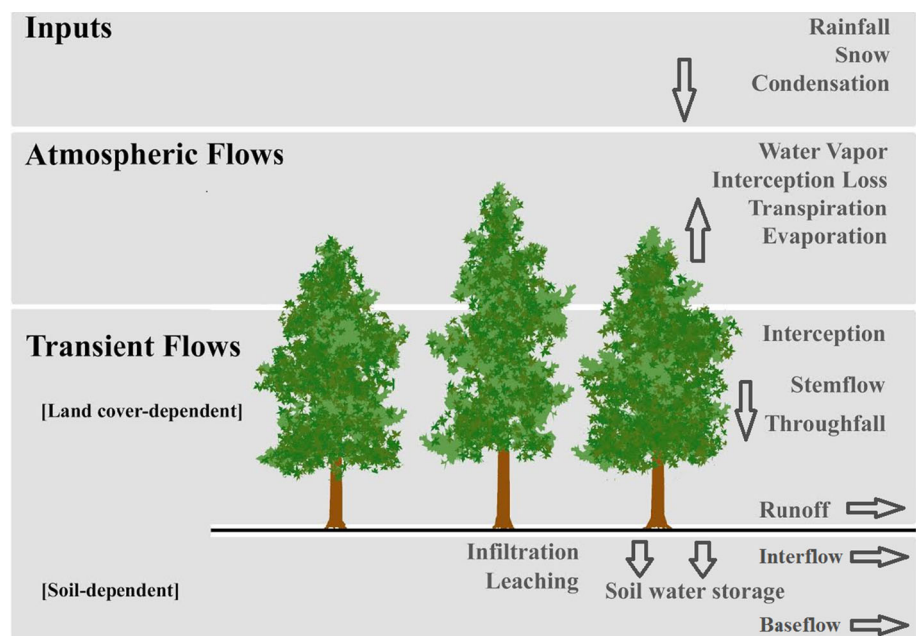
impact (Pielke et al. 2007; Neary and Koestner 2012). For example, plantations with planting densities and rotation times that are similar to previous forest harvesting practices in the region will have less of an impact on water quantity relative to higher density plantations on short rotations. The latter may increase ET, relative to the reference state.

The overall impact of woody bioenergy plantation systems depends on their impacts on both evaporation and transpiration. In temperate and tropical forests and plantations, ET can account for 55–95 % of gross precipitation, for annual rainfalls of 2500 and 500 mm, respectively (Zhang et al. 2001). In forests, rainfall interception loss accounts for 6–45 % of gross precipitation (Carlyle-Moses and Gash 2011) and accounts for the majority of evaporative losses. Canopy water storage is typically defined as the amount of water that can be stored in the canopy. Stemflow represents the precipitation that runs down the stem to the forest floor, which is often less than 10 % of gross precipitation (e.g., Levia et al. 2011). Transpiration rates vary among species and are influenced by environmental factors, plant anatomy, canopy structure, physiological properties, and seasonal changes in plant metabolic needs (Asbjornsen et al. 2011). For example, comparing among major woody bioenergy plantation systems, stand transpiration rates range from 180 to 400 mm/year for pine plantations (Putuhena and Cordery 2000; Licata et al. 2008) to 500–900 mm/year for many eucalyptus plantations (Putuhena and Cordery 2000; Lane et al. 2004; Almeida et al. 2007).

A myriad of environmental factors and management practices can affect the hydrologic impacts of woody bioenergy feedstock development. For example, meteorological conditions can explain a portion of rainfall interception loss, but the structure of the crop or forest also exerts considerable influence on the magnitude of rainfall interception loss (e.g., Licata et al. 2011; Pypker et al. 2011). Canopy water storage is influenced by leaf shape (Horton 1919), leaf surface features, such as hairs (e.g., Grah and Wilson 1944), and leaf configuration (e.g., Keim et al. 2006). Species with higher leaf area indexes provide more surface area, thereby increasing storage. In forests, canopies that support epiphyte communities or numerous dead branches provide increased storage that can often equal the canopy storage found on the trees themselves (e.g., Hölscher et al. 2004; Pypker et al. 2006). Moreover, in regions that experience frequent and persistent fog, tall stature vegetation can increase total water inputs via 'fog drip' (also referred to as 'horizontal precipitation') beyond rainfall inputs alone, a process that is enhanced by epiphytic cover, and can be lost with conversion to short stature vegetation (e.g., Bruijnzeel et al. 2011). Hence, any change in plant surface area or the amount of epiphytes or dead branches in the canopy will alter the canopy water storage. Canopies with larger gaps will allow for more direct throughfall to the ground and higher rainfall intensities at the soil surface.

While the quantity of stemflow per hectare is small (e.g., Levia et al. 2011), stemflow can concentrate inputs of water and nutrients toward the base of the plant. In forests, a change in tree structure directly influences the amount of stemflow. For example, smooth-barked species with erect branches tend to promote more stemflow, whereas rough-barked species with more horizontal branching tend to produce less stemflow (e.g., André et al. 2008). Therefore, changes in tree species can have dramatic impacts on the spatial pattern of precipitation below the forest canopy, influencing nutrient inputs, rainfall intensity, water infiltration, and subsurface water movement (e.g., Liang et al. 2009; Ford and Deans 1978).

Canopy structure also influences evaporation of water during a precipitation event (e.g., Carlyle-Moses and Gash 2011), which is often substantial, ranging from 0.4 to 25 % of gross precipitation (e.g., Link et al. 2004). Structural differences in the canopy will affect canopy conductance—the ease of water movement from the canopy to the atmosphere—with rougher canopies with high leaf areas near the top of the canopy likely having a larger conductance than a smooth canopy with most of the canopy water storage located closer to the forest floor (e.g., Pypker et al. 2011).

In sum, all of these components affect water fluxes between the atmosphere, vegetation, and soils and may need to be considered when assessing the potential hydrological impacts of woody bioenergy systems. More research is needed to determine which processes are most important and to understand how different bioenergy plantation systems—including different plant species, planting densities, rotation cycles, species mixtures, and management practices—vary in their ecohydrologic functions, which in turn will allow for better prediction of impacts on water resources at the stand to the watershed scale.

Landscape Scale Impacts on Flows

Concern has been expressed that biofuel production will compete for water resources with other uses, including food crop irrigation, in areas already under water stress (NRC 2008; Pimentel et al. 2008; Dominguez-Faus et al. 2009; Solomon 2010; Gerbens-Leenes et al. 2012). The impact of biofuel cultivation on water supplies is a function of water use by biofuel crops, which varies by plant and with climatic factors (e.g., precipitation and evapotranspiration rate), the occurrence of competing water users, and water availability (NRC 2008; Robertson et al. 2011). Most studies of water supply impacts use scenarios that prescribe crop mixes for a given region, based on policy-driven

targets for producing biofuels. The cropping scenarios, along with climatic forcings, drive agro-climatic models that provide estimates of water consumption. Water consumption is expressed as total rainwater or irrigation losses or in terms of water footprints (WF). Water footprints are split into green and blue fractions, which refer to water consumed from precipitation-derived soil moisture and from groundwater aquifers or surface water bodies, respectively.

Based on regional targets for biofuel production, Gerbens-Leenes et al. (2012) predict that in 2030, the global blue biofuel WF could grow to 5.5 % of the total available blue water. In India and China, De Fraiture et al. (2008) found that, in order to meet national policy goals for biofuel production, cultivation of sugarcane and maize would have to increase 16 and 26 %, respectively. The increase in crop cultivation would require corresponding increases in annual irrigation withdrawals of 35 and 30 km^3 in China and India, respectively, corresponding to 8 and 9 %, respectively, of total irrigation withdrawals in 2005. Yang et al. (2009) also estimated water requirements for China to meet policy goals, finding that irrigation requirements to meet 2020 biofuel production targets would be 32–72 km^3/ year, which relates to 7–15 % of total irrigation withdrawals in 2005.

Gerbens-Leenes and Hoekstra (2011) estimated that European Union (EU) targets for renewable transport fuels would lead to a WF of 62 km^3 per year, equivalent to 10 % of the current total WF of the EU. However, country-specific WFs vary an order magnitude, due to differences in use of transport fuels and potential biofuel production. In Spain, Elena and Esther (2010) found that national policy-driven biofuel cultivation scenarios would produce a 49 % increase in Spain's current agricultural WF. The simulated crop-biofuel system was a mixture of grain and oil seeds, used to produce ethanol and biodiesel. To meet the biofuel targets, the crops originated from imports or within the country based on current trade patterns. Wigmosta et al. (2011) calculated that meeting the US Energy Independence and Security Act (EISA) year 2022 biofuels production target with algae-derived biofuels would use 25 % of the current US irrigation demand. Potential locations of algae ponds were based on factors such as land use and land cover and topography. Yang et al. (2011) also estimated water consumption for algae-based diesel oil, finding that generation of 1 kg biodiesel requires 3726 kg water. They also calculated that, meeting EISA 2022 mandates with biomass-based diesel with algae cultivation would result in an increase of 9.7 % in water usage in the US.

Hernandes et al. (2014) evaluated the WFs of biofuels in south-central Brazil, finding that they range from 70 to 100 L/MJ for sugarcane ethanol and 40–50 L/MJ for biodiesel, depending on the region. Although the blue WF of biofuels is currently small, since biofuel crops are mainly rainfed in this region, total WFs can be reduced by increasing irrigation due to the resulting increase in productivity. However, this of course would increase the blue WFs of these biofuels. Since certain regions show better performance than others for converting irrigation water into yields (due to differences in soils, topography, etc.), the authors suggest that focusing growth in these regions could reduce biofuel crop expansion pressure in other regions; however, water availability for irrigation should be taken into account.

Chiu and Wu (2012) showed that WFs for corn grain, stover, and wheat straw can vary by more than an order of magnitude across US counties. Water demand for cultivating a typical mix of these feedstocks to meet EISA mandates would result in consumption of an equivalent of 0–21 % of local precipitation, depending on the county. Fingerman et al. (2010) assessed water consumption patterns associated with cultivation of ethanol-producing crops in California. Water consumption for ethanol production was estimated to be a thousand times greater than gasoline, on average, and predicted consumption varied by more than 350 % across the California counties.

The energy conversion side (ECS) of bioenergy production is generally characterized by smaller impacts on water availability than the feedstock production side (FPS), but this depends on the biomass and conversion process used, local conditions, and the scale of the assessment (Berndes 2002; Smeets et al. 2008). For example, a 400-million liter/year conventional dry-mill ethanol plant typically uses 1.2–2.4 billion liters of water per year, while a wet-mill ethanol plant uses even more (Pate et al. 2007), and this water consumption may have a significant impact on local stream flows and groundwater levels, but minimal impact at the scale of the FPS for the plant (equivalent to 12–24 mm/year over 10,000 ha). However, water consumption in the ECS is blue water, whereas in the absence of irrigation it is green water that is consumed in the FPS, which may be an important distinction in some watersheds (e.g., Yeh et al. 2011). Lignocellulosic ethanol production processes are still being developed, and water use and consumption rates for commercial operations are not fully established. Target projections are for commercialized enzymatic biochemical ethanol production to use on the order of 6 liters of water per liters of ethanol produced. The current state of the art with corn stover falls within the range of 10–11 liters of water per liter of ethanol. Current projections are that thermochemical approaches using water-based cooling systems will require twice as much water, with the 2012 water use target for woody feedstock to be about 12 liters per liter of ethanol (Pate et al. 2007; UNEP 2011). Since ECS water use efficiency is an immediate

concern to mitigate impacts on water resources at local scales, particularly in water-scarce regions, water research should be intensified for second-generation technologies.

Landscape Scale Impacts on Water Quality

Agriculture is already known to be a large contributor to excess nutrients, pesticides, and suspended solids to aquatic environments, and concern has been expressed that expansion of biofuel feedstock cultivation may exacerbate this problem (e.g., Pimentel et al. 2008; Schnoor et al. 2008; Solomon 2010). Most water quality-based analyses of biofuel cultivation focus on estimating loadings of pollutants to surface waters by feedstock cropping systems, i.e., feedstock crops and rotations. Hypothetical shifts from year-to-year corn–soybean rotations to more intensive corn-production rotations were found in a modeling study to produce greater losses of suspended solids, nutrients, and pesticides from agricultural fields (Thomas et al. 2009). Thomas et al. (2009), however, also found that best management practices could significantly offset these increases in losses. Perennial crops such as switchgrass and miscanthus are usually found to be less polluting because they require lower inputs of fertilizers and pesticides, in comparison to grain crops, per unit of net energy production (Tilman et al. 2006; Schnoor et al. 2008). Perennial crops tend to have greater soil and nutrient retention because of the continuous presence of subsurface biomass and longer periods between harvests (Robertson et al. 2011).

Dominguez-Faus et al. (2009) estimated that, on average, switchgrass will contribute lower loadings of nitrogen to streams than corn. Chiu and Wu (2012) use a gray-water footprint calculation approach to compare the vulnerability of streams across the U.S. to cultivation of conventional fuel feedstocks (corn grain and soybean) and cellulosic fuel feedstocks (corn stover and wheat straw). Gray-water footprints refer to the amount of water needed to dilute a pollutant to background levels, reflecting the local magnitude of the pollutant input, the local leaching potential for the pollutant, and the local background level of the pollutant (Hoekstra and Chapagain 2008). They found that corn stover has the highest gray-water footprint, on average, followed by corn grain, wheat straw, and soybeans, but that gray-water footprints for the four crops vary by orders of magnitude across the U.S., depending on climatic factors.

Shepard (2006) reviewed the best management practices (BMPs) to protect water quality during forestry operations that have been developed over the past several decades in the United States. These practices are being implemented routinely and have been shown to be effective in protecting water resources from potential impacts such as sediment and nutrients in runoff. State BMP manuals typically include sections on timber harvesting, site preparation, reforestation, stream crossings, riparian management zones, prescribed burning and fire lines, road construction and maintenance, pesticides and fertilizers, and wetlands. Thus, bioenergy production from conventional forestry should also be compatible with maintenance of high water quality.

Wu et al. (2012) used watershed hydrology and water quality models to examine the water quality impacts of modifications of corn-based cultivation practices, including increased corn yields, harvesting of corn stover, and incorporation of switchgrass in the Upper Mississippi River basin. Increases in corn yields would reduce nitrogen loads but slightly increase phosphorus loads. Use of corn stover and incorporation of switchgrass would reduce nutrients and incorporation of switchgrass would substantially reduce suspended solids loads. In a similar study in the Iowa River basin, Wu and Liu (2012) found that removal of corn stover for biofuel production would increase suspended loads greatly but reduce nitrogen loads. Conversion of even a small fraction (10 %) of cultivation area to switchgrass or miscanthus would substantially reduce sediment suspended loads.

Love et al. (2011) used a modeling approach to assess the effects of a range of hypothetical biofuel feedstock cultivation strategies on pesticide concentrations in streams in several large Michigan watersheds. They used miles of impaired stream reaches, with respect to human and ecosystem toxicity thresholds, as a measure of water quality impacts. Switchgrass and miscanthus cultivation scenarios had the lowest number of impaired stream miles, due to either lower required inputs of pesticides overall or application of pesticides that are less mobile and toxic. Similar results were found by Love and Nejadhashemi (2011) for nitrogen, phosphorous, and suspended solids, where loads for these pollutants were lowest for switchgrass and miscanthus rotation scenarios. These results were explained by perennial crops requiring lower nutrient inputs and producing less soil disturbance.

Using a water quality model, Ng et al. (2010) compared watershed nitrate loads generated from miscanthus cultivation against loads from corn–soybean crop rotations. Reductions in nitrogen loads were non-linearly related to increases in miscanthus cultivation, with higher fractions of miscanthus leading to proportionally lower reductions in nitrogen loads. Parish et al. (2012) used single- and multi-criteria simulation–optimization models to find optimal locations for switchgrass cultivation in a Tennessee watershed. The criteria included maximizing profits and minimizing pollutant loads. The various combinations of criteria generated similar results in terms of profits and pollutant loads, a result that was attributed to the small size of the potential area allowed for cultivation relative to the total watershed area. Gopalakrishnan et al. (2012) proposed

to use miscanthus and switchgrass as buffer strips between corn–soybean fields and watercourses. Using a biogeo-chemistry-crop yield model, they found that the buffer strips produced substantial yields of the biofuel crops while reducing nitrate leached to adjoining streams associated with fertilizers applied to the corn–soybean crops.

Along with the feedstock production side (FPS), the energy conversion side (ECS) of bioenergy production can have significant water quality impacts due to chemical and thermal pollution through the discharge of effluents (waste or co-products) into aquatic systems (Berndes 2008). Regardless of biomass used for biofuels, distillery wastewater, or stillage, is the most substantial by-product of the bio-mass-to-fuel conversion process (Wilkie et al. 2000; Fuess and Garcia 2014). Stillage characteristics are variable and depend on the biomass as well as several aspects of the production process, but in general, this residue presents high organic load values (biochemical oxygen demand, BOD) and can result in major environmental impacts, such as hypoxia. Gunkel et al. (2007) measured a BOD of 10,800 mg/L in sugarcane distillery stillage (vinasse) diluted for fertigation, and Moraes et al. (2014) report that approximately 10–15 liters of stillage are generated per liter of gross production of ethanol, thus, posing a serious waste disposal problem.

In northeastern Brazil, problems of land disposal of vinasse have already become evident, despite the prohibition of stream discharges and lagoon treatment (Christofoletti et al. 2013). For instance, all rivers in the coastal region of the state of Pernambuco are influenced by sugarcane cultivation (Gunkel et al. 2007), Effects are often magnified by dams, constructed for both urban water supply and electrical power generation. These reservoirs might also contribute to further water quality deterioration through eutrophication processes (UNEP 2011). According to Smeets et al. (2008), water pollution is also an important problem in some regions of south-central Brazil (e.g., in the Piracicaba River basin), but in most of these regions there are different water polluting sectors present, and there is insufficient information to determine the contribution of cane and ethanol production. For example, a recent study in São Paulo state (Guarenghi and Walter 2014) concluded that, based on the limited monitoring data available, it was not possible to determine the impacts of sugarcane expansion on water resources relative to the impacts of urban and industrial development.

Case Study: Great Lakes Phosphorous Loads

LaBeau et al. (2014) used a land-use prediction model coupled with a calibrated nutrient fate and transport model (SPARROW, Robertson and Saad 2011) to estimate increases in phosphorous loads to the Laurentian Great Lakes, as a result of expansion of corn cultivation for biofuel production. Predictions of phosphorous loads were made for 102 tributary watersheds in the US portion of the Great Lakes basin. Over the prediction period of 2010–2040, increases in biofuel feedstock cultivation are predicted to increase total phosphorous loads to the Great Lakes by 2.4 %. However, 12 of the 102 watersheds in the basin are expected to experience increases of 10–50 %; another five watersheds are predicted to have increases of over 100 %. Here, we analyze the results of LaBeau et al. (2014) to illustrate the influence of watershed geography on phosphorous loads. Four factors contribute to the variability in contributions of watersheds to phosphorous loads to the Great Lakes: (1) amount of land dedicated to corn cultivation within a watershed; (2) fertilizer and manure application rates per given area of cultivated land, which is a function of local soil quality and cultural practices; (3) local land-to-water delivery subsurface features, namely soil permeability and the presence of tile drains; and (4) stream hydraulic retention times, which depend on stream discharge rates, travel distances, and in-stream reservoir volumes.

Figure 3 shows rankings of the watersheds with the 10 highest predicted increases in phosphorous loads (kg/year) with respect to each of the four factors. Areas of the 10 watersheds ranged from 2000 to 13,000 km². The land-to-water delivery subsurface factor is calculated as the
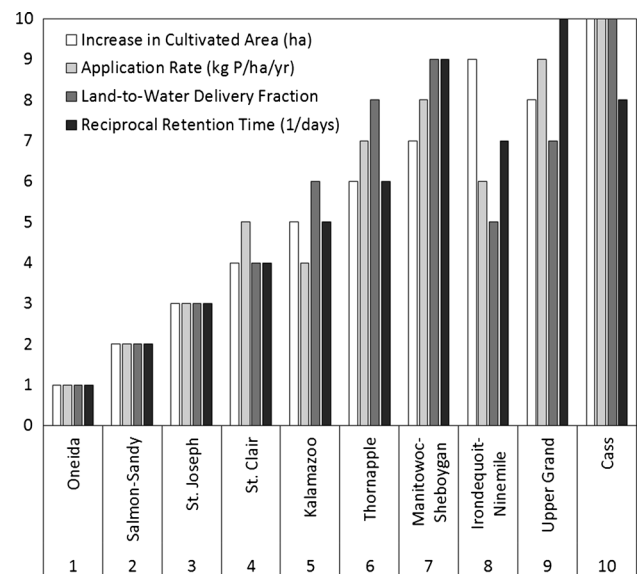
Fig. 3 Rankings of 10 Great Lakes watersheds with the greatest increases in predicted phosphorous loads (kg/year) with respect to increases in land dedicated to corn cultivation, fertilizer and manure application rates, land-to-water delivery fraction, and reciprocal stream hydraulic retention times. Ranks of watersheds in terms of phosphorus loads are indicated on the horizontal axis. The height of the bars indicates the rank of the watershed, among the 10 watersheds, with respect to each factor

fraction of applied phosphorous that is transported from the land surface to adjacent streams, which is calibrated based on soil permeability and fraction of land underlain by tile drains. The hydraulic retention time is inverted so that high values represent greater delivered loads. The 10 watersheds listed in Fig. 3 account for 90 % of the total predicted increases in tributary watershed loads. The rank of the increase in cultivated land area parallels the overall rank consistently, indicating that this factor is the most important determinant of increase in phosphorous load. Reciprocal retention time roughly corresponds to the overall rank, indicating that this factor is also important in determining load increases, but is secondary to increases in cultivated area. In other results from LaBeau et al., this factor is also shown to be especially important as a limiting factor, where some watersheds with high increases in cultivated land, but high retention times due to distance or the presence of significant reservoir storage, had low increases in the amount of phosphorous delivered to the Great Lakes. The land-to-water delivery factor rank is inconsistent with respect to overall rank, because soil permeability tends to be similar and tile drain coverage is low for the 10 watersheds. Finally, application rates are also similar among the watersheds, leading to a lack of correlation between this factor and the overall ranking.

The importance of each factor in the phosphorus loadings in this case study is specific to the region and the predicted biofuel cultivation expansion scenario. However, the results imply that geographic differences in the intrinsic features of the watershed (e.g., soil types, artificial drainage systems, hydraulic residence times to sensitive water bodies) can be important for determining nutrient or other pollutant loads emanating from watersheds with biofuel crop cultivation. These factors should be considered in situations where pollutant loads associated with biofuel feedstock cultivation are to be minimized on a regional basis.

Managing Impacts of Biofuel Feedstock Cultivation and Processing

Species Selection and Combinations: Physiological and Genetic Characteristics

Responsibly using water resources presents new challenges and opportunities for selecting species with particular physiological and genetic traits that optimize the balance between productivity and environmental impacts in bioenergy production systems. For example, a physiological trait that can directly influence water use patterns is water-use efficiency (WUE), the ratio of the amount of water transpired to the amount of carbon fixed via photosynthesis, which can vary greatly among species growing under similar environmental conditions (e.g., Dierick and Hölscher 2009). Improving water-use efficiency (WUE) of the crop or forestry system used for bioenergy production is essential to reducing the total transpired water by the bioenergy crops. Rain WUE could be improved by reducing rain interception losses, increasing the leaf-level WUE, or through changing allocation partitioning toward the harvested product (Condon et al. 2004). Selecting genotypes with morphological traits that reduce rainfall interception loss offer opportunities for improvement in the WUE of woody species, as the impact on rainfall interception loss could be larger than in shorter crops (Calder 1998).

Moreover, while there are usually tradeoffs between leaf-level WUE and productivity, the morphological characteristics determining rainfall interception loss may not be related to physiological productivity traits. It also represents a new challenge to quantify and select genotypes in traditional breeding trial plots, as the gain from any particular genotype because of reductions in rainfall interception loss will be distributed among the neighbors. In other words, neighboring plants and plots could be growing faster because of the extra water that a low rainfall interception loss genotype is letting through. Aggressively competitive genotypes would exacerbate this behavior. It would be important to test individual genotypes in large plots in order to be able to measure the gain in productivity due to lower rainfall interception loss. In contrast, identifying morphological traits that do not reduce productivity and at the same time reduce rainfall interception loss (e.g., upward branching species, smooth bark and leaves, and lower LAI Pypker et al. 2011) would allow for a creation of gradable morphological indexing of genotypes that can be assessed together with productivity traits.

The increasing need for superior genotypes, along with the development of affordable new generation sequencing technologies, has led researchers to explore technological alternatives to increase biomass production down to the cellular level (Porth et al. 2013; Rogers et al. 2012). In crops, stronger economic incentives and a faster crop rotation time have led to a further understanding of their genetic make-up (Hufford et al. 2012; Schnable et al. 2009) and their implications for bioenergy production (Tyndall et al. 2011; Zhuang et al. 2013). Despite the benefits of sequencing technologies, the expansion of genetically engineered tree species for bioenergy production has been slowed down, in part due to their longer life cycles and a persistent negative social acceptance of genetically engineered trees (Flachowsky et al. 2009; Valenzuela et al. 2006; Whetten and Kellison 2010). Despite these limiting factors, research has been conducted to develop highly productive and frost resistant eucalyptus hybrids

(*Eucalyptus grandis* × *Eucalyptus urophylla*), characteristics acquired with the introduction of a plant transcription factor that regulates the cold-response pathways (Hinchee et al. 2009). Similarly, Fang et al. (2013) successfully tested biomass production properties of 25 hybrid poplars and concluded that hybrids have indeed more advantages over traditional poplars in terms of bioenergy production. While trials on hybrid species to determine biomass yield and energy values are important (Pilate et al. 2012; Zamora et al. 2013), research has also been conducted to assess ecological impacts of large-scale plantations of hybrid poplars (Zalesny et al. 2012), which has the potential to define suitable lands for poplar production systems.

Impacts of Feedstock Cultivation Management Practices on Field Scale Water Budgets

Regardless of the species planted, changes in management practices will directly impact plot-scale water budgets. Any management that reduces leaf surface area will reduce canopy water storage and, if gaps are created in the canopy, increase direct throughfall. In regions with little fog, the immediate impact will likely reduce interception loss, but the impact is not likely to be linearly related to the loss of biomass. Changes in the stand canopy will alter canopy conductance (g_a). If whole trees are removed, g_a will likely decrease for the stand but will increase for the trees near the new opening (e.g., Teklehaimont et al. 1991). In regions that consistently have fog, impaction by fog on the trees can result in negative rainfall interception loss (e.g., Crockford and Richardson 2000). Loss of the trees in fog-dominated systems would decrease water inputs to the soil. Hence, there is a complex interaction between the physical structure of the stand, canopy, meteorological conditions, and interception loss.

In woody biofeedstock production systems, silvicultural management will also play an important role in determining whether bioenergy feedstock production will change water flows in the forest landscape, and if so, how. While short rotations can lead to reduced rates of transpiration, high density plantations can have the opposite effect (Kauter et al. 2003; Dillen et al. 2013). Research on the hydrological impacts of bioenergy production has mostly focused on small stature (mostly annual) crop species (e.g., Stone et al. 2010; Basso et al. 2012). A few studies have considered hydrological implications of taller stature (perennial crop) bioenergy plantations, including palm plantations (Madurapperuma et al. 2009; Babel et al. 2011) and fast-growing tree species (Allen et al. 1999; Sochacki et al. 2007; Neary and Koestner 2012). However, there is still a knowledge gap of how bioenergy-oriented forestry will impact water use at the tree and stand levels.

Bioenergy feedstock production systems are typically managed as short-rotation crops, maintaining a young cover of trees with high growth rates and often under a coppice regeneration system with species such as hybrid poplars (*Populus* spp.), willow (*Salix* spp.), sycamore (*Platanus* spp.), and silver maple (*Acer saccharinum* L.). Studies under a variety of environmental conditions have reported gains in biomass production of up to 12 tons of dry matter per hectare per year (e.g., Kauter et al. 2003; Zalesny et al. 2012; Dillen et al. 2013). However, high growth rates generally lead to higher stand water use compared to native forest ecosystems or plantations managed for timber production (e.g., Dimitriou et al. 2009; Forrester et al. 2010; Aranda et al. 2012). Water-use patterns can vary widely among different species typically used in plantation systems (e.g., Dierick and Hölscher 2009; Kunert et al. 2012). A few studies have specifically assessed bioenergy plantations and have found that exotic, hybrid, and/or fast-growing species are generally more productive and transpire greater amounts water than native forests (Licata et al. 2008; Kagawa et al. 2009; Larcheveque et al. 2011), and several recent studies have cautioned about the potential for high water use by hybrid poplar plantations (e.g., Dimitriou et al. 2009; Petzold et al. 2011; Schmidt-Walter and Lamersdorf 2012). Schmidt-Walter and Lamersdorf (2012) also point out that proper management practices can mitigate these water-use impacts, and there may be positive impacts on water quality. Screening tree species using a range of ecophysiological measurements and indices therefore offers a promising approach for assessing potential tradeoffs between production and water use, and selecting appropriate species to best meet the management goals (Wang et al. 2013).

Another area of growing interest has been the hydrological tradeoffs between monoculture versus mixed-species plantations. To date, much of this work has focused on plantations established for timber production or ecological restoration (e.g., Grossiord et al. 2013; Forrester 2014); however, there are important implications for bioenergy production that will likely demand greater attention in the future. In particular, mixed-species plantations are thought to increase stand productivity over monocultures by creating opportunities for complementary (obtaining resources from distinct pools in space or time) or facilitative (processes by which one species enhances the availability of a resource to other species) interactions among species with differing ecological strategies, thereby leading to greater total resource use (Binkley et al. 2004; Jose et al. 2006; Kelty 2006).

Several studies have documented higher productivity for mixtures than for monocultures in timber plantations (Forrester et al. 2006; Potvin and Gotelli 2008) and in grassland systems (Tilman et al. 2006; Mangan et al. 2011,

but see Johnson et al. 2010 and Wang et al. 2010). Higher productivity in mixed-species plantations has also been associated with greater stand transpiration (Kunert et al. 2012); however, higher water-use efficiency in mixed-species stands may partially compensate for this tradeoff (Forrester et al. 2010). Comparable studies of the hydrological impacts of monoculture and mixed-species of short-rotation bioenergy production systems are lacking and represent a critical area for future research.

Hydro-economic Modeling for Integrated Management

Replacing the traditional, fragmented approach to water resources management with a more holistic approach will be essential to understanding and managing several unique aspects of bioenergy-water resource systems. One unique aspect of these systems, compared to other agricultural systems, is the potential for conflicts stemming from food-energy tradeoffs and water allocation (e.g., water use for bioenergy feedstock production and processing rather than food crops or other energy sources, such as hydroelectric energy). Fragmented analyses may fail to capture important aspects of these tradeoffs. For example, in the U.S., not all corn diverted to ethanol needs to be replaced because an ethanol by-product, dry distillers' grains, replaces roughly one-third of the animal feed, with indirect reductions in impacts on water resources (Searchinger et al. 2008).

Another unique aspect of bioenergy systems arises, in many cases, due to the need for relatively short feedstock haul distances to ensure economic viability of biofuels (Wakeley et al. 2009). When biofuel processing plants are located close to feedstock cultivation areas, nonpoint source pollution from land-use change and point source pollution from biofuel processing can affect the same watershed. In contrast, this proximity of impacts can also create unique opportunities for economic recovery of wastes, and additional profits, through integrated management of ECS and FPS processes. For instance, the use of vinasse as a fertilizer in Brazil increases sugarcane productivity and reduces stillage treatment costs), but as mentioned previously, the reuse of vinasse has significant water quality impacts. On the other hand, some bioenergy supply chains may include longer transportation distances, especially if they are part of integrated food and/or feed production systems. Studies of long-distance supply chains include soy biodiesel and switchgrass pellets originating in Argentina (Van Dam et al. 2009) and cellulosic bioethanol in the U.S. (e.g., Huang et al. 2010; Bowling et al. 2011). In these systems, water resources impacts would likely be regional in nature, i.e., spread across multiple watersheds.

Ideally, reuse of bioenergy by-products should generate economic value and reduce adverse environmental impacts from the entire bioenergy chain. Focusing on water availability impacts, for example, the use of by-products for process heat or additional fuel and electricity production can substantially reduce water consumption per GJ bioenergy (Berndes 2002). Electricity generation from biogas (produced from anaerobic digestion of vinasse in Brazil), instead of using natural gas, has the potential to reduce CO_2 emissions by 0.8–1.2 ton/m^3 of ethanol production (Van Haandel 2005). Furthermore, the biogas (methane) option is attractive since it concentrates the dry solid content of the vinasse, thereby reducing the BOD and adverse environmental impacts of its land disposal. Since the process does not remove nutrients such as potassium, the residue can continue to be used as fertilizer. However, anaerobic digestion is expensive and there is not a significant volume reduction, meaning that transportation costs would remain high. Thus, concentration of vinasse by evaporation for the production of potassium fertilizer may be beneficial both in terms of profit and reducing environmental impacts compared to fertirrigation with high transportation costs (Larsson and Tengberg 2014).

Industry is likely to adopt or develop integrated technologies when each by-product in the production chain generates economic value at minimum environmental impact (i.e., investment risk due to environmental regulations) (Wilkie et al. 2000). In this context, hydro-economic models can be a useful tool to estimate abatement costs, damages from ambient pollution, direct economic benefits, and societal benefits, among others, in support of decisions for improved joint water quantity–quality management in bioenergy systems. Hydro-economic models have long provided a conceptual platform for integrated water resources management on regional scales (e.g., Bear et al. 1964; Gisser and Mercado 1972; Noel et al. 1980). These optimization models prescribe strategies for water allocation from an economic point of view, while incorporating hydrological and environmental constraints (Harou et al. 2009). In recent years, model scope has broadened to incorporate institutional and policy constraints, in order to evaluate the impacts of such constraints (e.g., Mayer and Muñoz-Hernandez 2009).

Two basic approaches to integrating economic and hydrological components in optimization models have been termed "compartmental" and "holistic" modeling. In the former, hydrological and economic components are developed as separate sub-models whose individual solutions are modified by a coordination method. Although this can facilitate model development—essentially the sub-models can be developed in parallel by separate teams of analysts, each with specific expertise—it can be difficult to integrate them effectively (Cai et al. 2001). In the latter approach, model components are closely connected in such a way that physical and information feedbacks occur between them

endogenously. The endogenous treatment of hydrological and economic relationships has several advantages, including allowing for a more detailed representation of human-environmental feedbacks and hence a more effective analysis of decisions related to a sustainable economic development (Cai et al. 2008). The holistic hydro-economic models have proven particularly useful for regions where competition for water is fierce, where economic uses of water dominate, and where sufficient hydrologic and economic data are available to calibrate the models (Marques et al. 2006).

These conditions for hydro-economic modeling are all met in some basin rivers in Brazil where sugarcane-based ethanol is produced. Two example applications illustrate the use of integrated hydro-economic modeling for developing policies that promote sustainable resource use and economic development at the river basin scale. The first case study assesses the impacts of policies on nonpoint source pollution and water quality outcomes in the Pirapama River basin in northeastern Brazil (Moraes et al. 2010), and the second case study focuses on integrated water-energy management in the Araguari River watershed in southern Brazil (Marques et al. 2010). A third case study presents an economy-wide global model that investigates the potential for biofuels feedstock production in Brazil considering the competition between food crops and sugarcane for limited land and water resources (Carneiro et al. 2014).

Applications of Hydro-economic Modeling in Brazil

In Brazil, energy from sugarcane biomass has surpassed hydropower generation, representing 15.4 % of all energy produced in 2012 compared to 13.8 % from hydropower and 39.2 % from petroleum (EPE 2013a). In terms of final energy consumption by source, ethanol accounts for 4.2 % and sugarcane processing by-products 11.2 % (EPE 2013a). This represents 23.5 billion liters of ethanol produced in 2012 (EPE 2013b). However, Milanez et al. (2012) contend that ethanol production in Brazil is not keeping up with demand, with a potential cumulative deficit reaching 32.4 billion liters by 2015.

While sugarcane is still largely produced without irrigation in Brazil, significant productivity gains are possible with irrigation (Dalri et al. 2008; Silva et al. 2014), potentially making sugarcane economically attractive in larger regions of the country. Machado et al. (2002) computed internal rates of return of up to 61.4 % for drip-irrigated sugarcane, compared to 26.9 % for non-irrigated sugarcane. Studies have shown that yields in the state of Piaui, in northeastern Brazil, can be more than doubled with irrigation (Andrade et al. 2009), and more and more of this region is being converted to irrigated area (Herreras

Martinez et al. 2013). In western São Paolo, where growing demand for sugarcane in south-central Brazil has led to exploitation of drier regions, studies have shown that drip irrigation is economically viable under certain conditions (Smeets et al. 2006). At a global scale, growing concerns about energy security and climate change have led to various regulatory policies and mandates in many countries to promote a transition to renewable energies, including liquid biofuels. As the world's second largest producer/consumer and the largest exporter of ethanol, Brazil plays a major role in the global biofuel economy. It is often argued that with a more efficient utilization of its vast amount of pasture lands, Brazil could expand its biofuel production further and to help meet the demand for sugarcane ethanol in the domestic and international markets.

While the issue of land-use change in Brazil has been addressed in several previous studies (Nuñez 2012; Elobeid et al. 2011; Nassar et al. 2009), less attention has been given to water resources. About 29 million hectares (ha) of the current total cropland used in Brazil is suitable for irrigated agriculture. However, as of 2007, irrigation infrastructure was developed on only 5.4 million ha of land (ANA 2012), where approximately 24 billion m^3 of water was used for irrigation, more than 14 billion m^3 of which was used by sugarcane (Carneiro et al. 2014, based on FUNARBE 2011). Therefore, an expansion of sugarcane acreage could pose serious challenges due to the high water-use intensity of this crop and limited water availability for irrigation. Increased water scarcity may also adversely affect other energy sources, including hydroelectric.

Case Study: Pirapama River Basin, Pernambuco State

All rivers in the coastal state of Pernambuco, in northeastern Brazil, are influenced by sugarcane cultivation, and stillage fertirrigation has been practiced in the region since 1981, when a state law prohibiting the direct disposal of sugarcane wastewater in surface waters was introduced. These rivers provide water supplies to urban areas, and water quality effects are often magnified by multiple dams for both water supply and electrical power generation (Gunkel et al. 2007).

Moraes et al. (2010) developed an integrated economic-hydrologic model to analyze joint water quantity–quality management associated with bioenergy production in the Pirapama River Basin in northeastern Brazil (Fig. 4). The Pirapama River is the most important water source for the city of Recife, capital of the state of Pernambuco. Water management in the basin is affected by both scarce water resources and regulatory limits on water pollution. To resolve domestic water supply problems, in 2008, a reservoir was built on the Pirapama mainstem, at a total cost of about US $250 million. While the reservoir is already producing
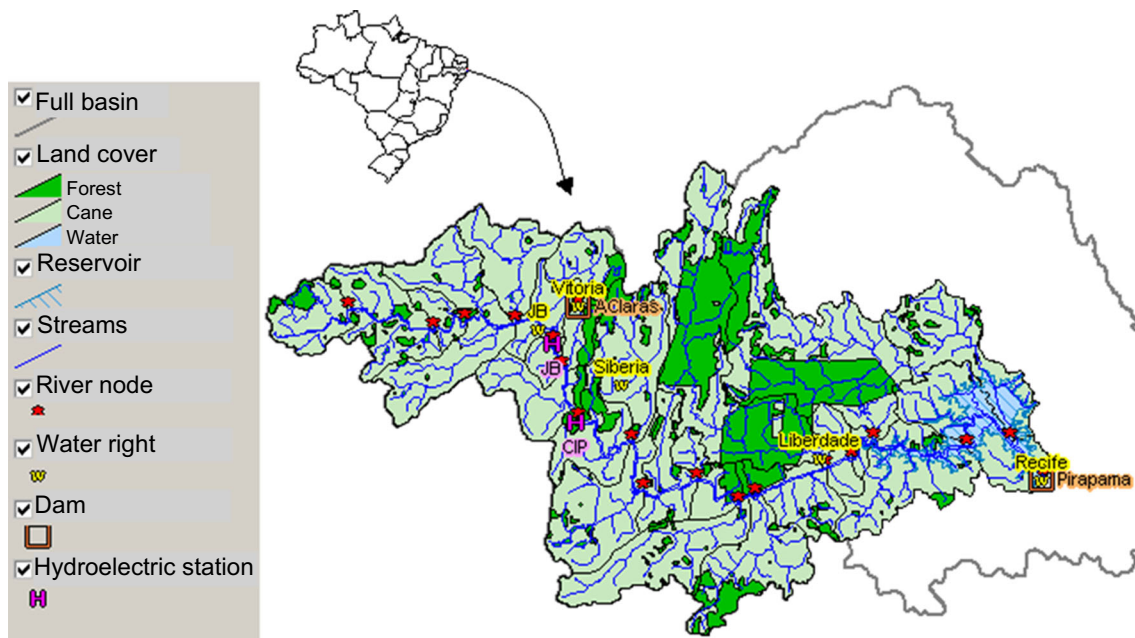
Fig. 4 Location of the state of Pernambuco and the Pirapama River Basin as well as the first stretch of the basin where the model is applied. *Source* Moraes et al. (2010). Used with permission from ASCE

around 40 % of the water used by Recife's metropolitan area, it may also exacerbate eutrophication and water quality degradation if no measures are taken to mitigate the impact of upstream sugarcane cultivation and ethanol production. Some studies (Rodella 2014) have estimated that annual water treatment costs will increase by approximately US $2 million if the Pirapama reservoir becomes eutrophic.

The integrated model by Moraes et al. (2010) includes bioethanol production facilities and the areas planted with fertirrigated sugarcane, with effluent discharge from the irrigated land assumed to be proportional to fertirrigated areas. A simplified riverine water quality model is also included. The integrated model is capable of (i) estimating direct economic benefits of bioethanol production and associated environmental impacts of different water allocation levels; (ii) assessing the economic effects of different limits on water quality constituents and ecological flows; and (iii) obtaining shadow prices that can be used to infer the cost of pollution prevention and design control measures or economic incentives, such as pollution taxes. Model results show that without water quality constraints, almost all water demands can be met, even during harvest months. However, if water quality restrictions are introduced, net benefits are reduced significantly for fertirrigation, even in a normal hydrologic year. Net benefits also decline for other water users as water quality restrictions require higher in-stream flows for waste assimilation and increased reservoir storage to avoid eutrophication. Results show further that economic

benefits for the agro-industries decline while social costs increase when water quality standards are met. To meet water quality standards, most of the region's agro-industries no longer allocate effluents to sugarcane areas; however, they face higher effluent transportation costs as a result.

More recently, the same framework was used to calculate how much transportation costs would have to decrease in order for the agro-industries to economically send these residues to more distant sugarcane cultivation areas. Reduction in transportation costs was shown to be more important than reduction in water treatment costs for improved distribution of effluents to the areas that cause less environmental damage. Also, new stillage handling options, such as anaerobic digestion and biogas production, were added to the model. It was shown that when existing water quality limits are enforced, the agro-industries would obtain greater net benefits by adopting this means of residue reuse. In dry years, the impacts of water quality constraints are more severe, as expected, which is a major concern given evidence of increased climate variability and climate change. However, new forms of vinasse disposal can help to mitigate the economic and environmental impacts of lower water availability in the Pirapama River basin (Germano 2011).

Case Study: Araguari River Basin, Minas Gerais State

The 22,086 km² Araguari River Basin (Fig. 5) is part of a larger region in Minas Gerais state that has undergone

Fig. 5 Study region, with location of main sugarcane processing plants (*triangles*). *Source* Marques et al. (2010). Used with permission from ASCE
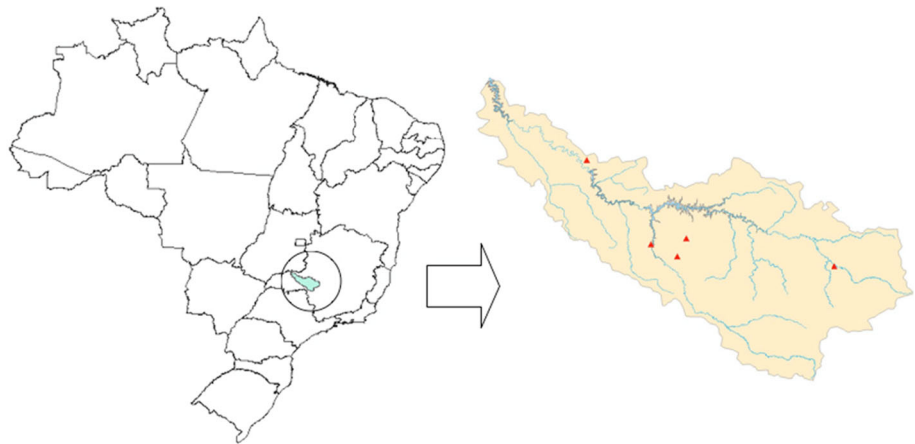
significant land-use change in recent years. The sugarcane area in the state increased from 226,400 ha to 495,900 in 2008 (Brito and Reis 2012) and was recently reported at 781,920 ha, with over 70 % of this production in the Araguari River basin (CONAB 2013). The river basin has two surface water reservoirs, including a large reservoir (12.7 billion m^3), and three run-of-river power plants with over 1,400 MW installed hydroelectric generation capacity, as well as other economic demands, including urban, industrial, and other crops. With rapidly growing sugarcane production, there is a potential conflict between ethanol production and hydroelectric energy generation.

To study the effects of increased ethanol production, a hydro-economic optimization model is developed to prescribe water allocations and surface reservoir operation at the basin scale. The model's objective function minimizes the water scarcity cost to various users, subject to physical and legal constraints in the system (e.g., Medellín-Azuara et al. 2009; Jenkins et al. 2003). The scarcity cost functions are based on water demand functions, developed with a crop optimization model that maximizes agricultural profit subject to constraints on water, capital, and other inputs, as in Howitt et al. (1999).

In the model, the Araguari River watershed is divided into ten sub-watersheds, each with monthly varying water scarcity functions reflecting sugarcane and other crops. Urban and industrial uses are assumed to have fixed water demands, and each sub-watershed is assigned an energy demand, to be met either by hydropower or ethanol (Fig. 6). The model evaluates tradeoffs associated with different water and energy allocations (hydropower plus ethanol) to meet renewable energy requirements. The model also allows the investigation of the effects of reducing water demands in urban regions (e.g., conservation measures) and reallocating the water to ethanol production, which will supply part of the urban energy demands with a cleaner fuel. Further, the model evaluates impacts on other
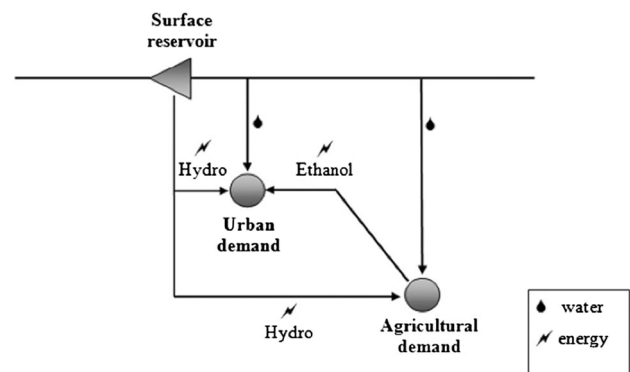
Fig. 6 Water and energy allocation. *Source* Marques et al. (2010). Used with permission from ASCE

crops competing for water and also on reservoir operation for hydropower. Overall, the integrated optimization approach supports adaptive measures in watersheds where the production of ethanol is expected to increase, as well as market-based solutions for efficient allocation among competing water and energy sectors.

One policy scenario evaluated by the model is the end of the gasoline subsidy in Brazil, which has hindered ethanol production. The model predicts that ending this subsidy would result in an increase in sugarcane production in the Araguari watershed of 3.14 million tons, and an increase in the planted area of 23,776 ha (4.3 % net increase), based on cross-price elasticity information (Orellano et al. 2014; Santos 2013). If irrigated, this increased area would boost water demand by 181 million m^3/year (approximately 24 %), assuming that 18 % of the current planted area is irrigated. Results from the hydro-economic model are useful for (a) evaluating the potential increase in water scarcity cost resulting from this scenario, (b) supporting irrigation policies across the watershed (i.e., where to expand based on water availability and scarcity cost), and (c) developing adaptation measures including reservoir

reoperation and conjunctive use of surface and ground-water to accommodate new demands, based on water costs, availability, and economic value.

Case Study: Economic Analysis of Land-Use Change and Irrigated Biofeedstock Production in Brazil

To investigate the potential effects of scarce water resources on sugarcane-ethanol production, Carneiro et al. (2014) extended the economic model of Nunez et al. (2013) to consider the competition for limited resources between sugarcane and several other major crops, all produced under restricted irrigation water availability in Brazil. Nunez et al. (2013) simulated the agricultural and transportation fuel sectors and determined the equilibrium in commodity and fuel markets while considering global commodity trade, without taking into account limited water availability for irrigation. The model by Carneiro et al. (2014) includes a water availability constraint. As a "high level' hydro-economic model, it does not explicitly consider green–blue water links, but it uses irrigation requirements from FUNARBE (2011) based on studies of the balance between precipitation and evapotranspiration for each crop, month, and region. Expansion of biofuel production is constrained by irrigation water availability, under the simplifying assumption that the percentage of irrigated area will remain constant in each region (i.e., if currently 10 %, then 10 % of the new area will be irrigated). In future climate scenarios, the irrigation water requirements are updated to reflect IPCC (2007) projected temperature and precipitation changes. Irrigation areas are also projected to expand in some regions based on public investments currently being made to expand irrigation infrastructure. In the simulation analysis, irrigation water availability is projected to be 25.6 million m^3 in 2030, based on the existing infrastructure and planned development projects.

Model results indicate that, without water limitations, 18.5 million ha of additional land would be allocated to crop production (43 % increase) by 2030, most of which (16 million ha) would come from the conversion of good-quality pastures to cropland, 53 % of which is used for sugarcane production. When water availability is incorporated, however, the outlook changes dramatically. Lands allocated to corn and soybeans increase slightly, and the sugarcane area is reduced significantly (approximately 12–25 % based on preliminary sensitivity analyses), mainly due to the limited water availability in regions that are most suitable for sugarcane production. The scenario of limited irrigation water favors corn and soybean production because sugarcane is a more water-intensive crop.

The results of this model indicate that an export demand shock for sugarcane would put pressure on other users, notably other farmers faced with increased scarcity of irrigation water (e.g., Torres et al. 2012). Green–blue water links, not explicitly considered in the model, could exacerbate these impacts. This raise questions about the ability of Brazil to meet future domestic and global demand for biofuels without integrated river basin management. Hernandes et al. (2014) proposed that investment in irrigation in regions where water resources are available can reduce land expansion pressure in other regions where land and water resources are scarce. However, Herreras Martinez et al. (2013) showed that for Northeast Brazil, and the entire country, the best socio-economic scenario may be one that considers the Northeast's more efficient agricultural practices and processing facilities, in addition to expansion of the sugarcane-ethanol sector into new areas. With several federal government initiatives aimed at increasing or revitalizing irrigated area in Northeast Brazil (Brazil 2012), including diversion of the São Francisco River to provide water to drier parts of the region (Brazil 2014), there are concerns about sustainability. For example, irrigation is already responsible for most of the water usage in the São Francisco basin, but it contributes only 11 % of the total water fees (ANA 2012). The price for the irrigation sector is much less than for other water users, which could lead to overexpansion, greater water scarcity, and conflicts in the future (e.g., Mirchi et al. 2012). Thus, hydro-economic model results based on various water availability scenarios may provide useful insights to agricultural producers and public policy makers.

Conclusions and Research Recommendations

The hydrologic impacts of bioenergy systems are dependent on a large number of factors, including the feedstock species, environmental conditions, management practices, and the energy conversion process. With respect to individual species, opportunities exist for increasing water-use efficiency through genetically improved varieties. At the plot scale, where there is a complex interaction between the physical structure of the crop or forest stand, meteorological conditions, and hydrologic processes (e.g., interception loss, throughfall, and evapotranspiration), opportunities exist for improved silvicultural management techniques and cropping systems. This indicates the potential benefits of extension services. However, additional research is needed to better understand hydrologic impacts and tradeoffs of the many different forest-based bioenergy plantation systems, particularly monoculture and mixed-species of short-rotation systems at high densities.

Water quality impacts of bioenergy systems are expected to be similar to those of other agricultural systems, where it has been shown that best management practices

can greatly reduce adverse impacts such as eutrophication resulting from increased runoff of nutrients. Cumulative impacts of large-scale bioenergy development at watershed or regional scales may be difficult to avoid, however, due to spatial variability in factors affecting pollutant transport to affected water bodies (e.g., soils and drainage networks). In a case study of potential phosphorus loading to the Great Lakes under a bioenergy expansion scenario, the total phosphorous loads to the Great Lakes were projected to increase by only 2.4 %; however, over 90 % of the increased loading was predicted to come from less than 10 % of the watershed area. This spatial concentration of the increased loads was due to a combination of some watersheds being more attractive for expansion of biofuel crops and intrinsic features of watersheds that lead to reduced attenuation of nutrient loads.

Driven by incentives to maximize profits and minimize environmental risks, industry has developed a more integrated approach to water and energy management. In some cases, energy conversion effluents can be used for fertirrigation of bioenergy feedstocks, although not without water quality impacts. In other cases, by-products can be used directly to produce additional energy, such as biogas. For example, in Brazil, sugarcane distillers use energy from the bagasse not only for their own needs but also to feed the power grid (Smeets et al. 2008), and this cogeneration of electrical power reduces GHG emissions and the overall water footprint of the sugarcane-produced ethanol.

Although integrated management approaches can significantly increase water-use efficiency and reduce water quality impacts, bioenergy production in many locations is expected to be limited by water resource constraints in the near future. Forward-looking studies (e.g., Berndes 2002) have shown that in water-scarce countries, water shortages will increase rapidly in scale and intensity even without new large-scale bioenergy feedstock production. Although we would argue that country-level analyses are often not relevant for policy making, because water scarcity can vary significantly on regional scales, watershed-level analyses in Brazil have confirmed that water availability appears not to impose a constraint on the assumed level of bioenergy production per se. However, water quality regulations would likely impose constraints if enforced, and increased irrigation of sugarcane would increase water scarcity and its economic value. In this context, it becomes critical for Brazil and other bioenergy producing countries to evaluate how their different watersheds can accommodate the growing water demand for bioenergy production, in order to avoid transferring the environmental and economic impacts of meeting international bioenergy mandates to the producing regions, potentially aggravating already complex water resources problems.

Recently, an economy-wide global model of the agricultural and transportation fuel sectors of Brazil and the U.S. (Nunez et al. 2013) was used to analyze the potential for expansion of the sugarcane-ethanol industry in Brazil. Results indicated that consideration of regional irrigation water constraints would significantly affect land use and livestock intensification, as well as limit the economic viability of expanding sugarcane production due to water scarcity costs (Carneiro et al. 2014). Investment in irrigation and regional infrastructure development would allow for greater expansion, but more water demand for irrigation would put pressure on other uses, which could intensify conflicts in water-scarce regions such as Northeastern Brazil.

Given the potential large-scale expansion of bioenergy, including large impacts on water resources and direct and indirect impacts on rural economies, it is important that water-use policies and instruments for bioenergy must be evaluated not only in relation to their effectiveness and efficiency—the focus of most studies to date—but also with respect to their socio-economic impacts (Moraes et al. 2011). New integrated modeling approaches are increasingly equipped to address these issues, for example, by modeling market-based approaches rather than command-and-control decision making (e.g., Bhatia et al. 2006, Zhao et al. 2009). Therefore, water policies related to bioenergy production should be evaluated based on measures of direct and indirect economic impacts, impacts on all water users and uses (including the poorest and most vulnerable segments of the population; e.g., Bhatia et al. 2006), and to the extent possible, backward and forward linkages related to both inputs and outputs of the bioenergy lifecycle (Moraes et al. 2011).

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