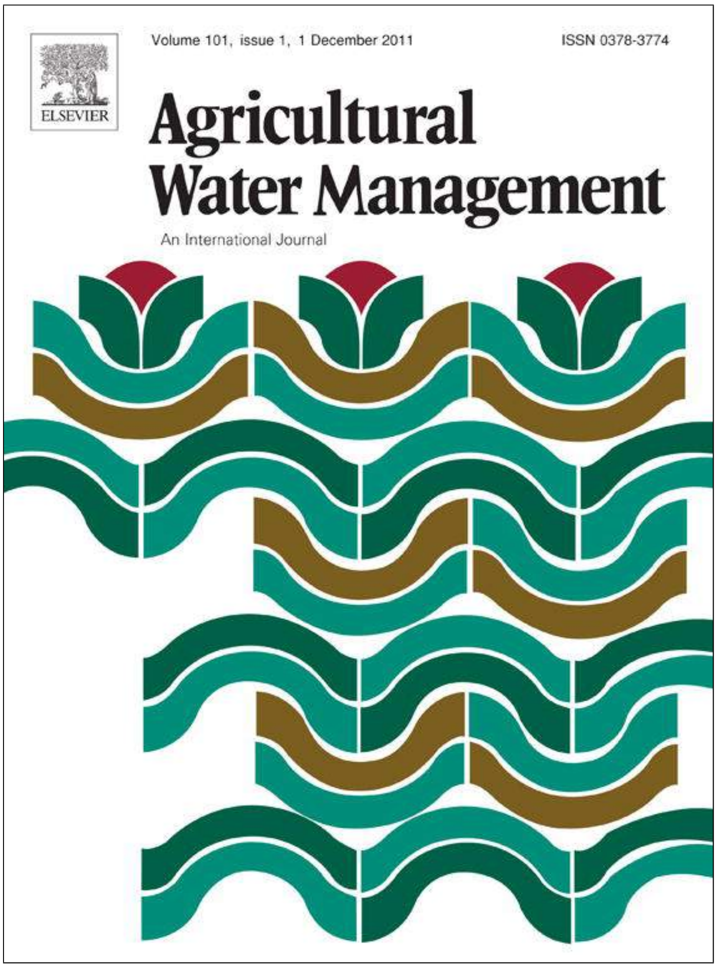
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Hydrological impact of biofuel production: A case study of the Khlong Phlo Watershed in Thailand

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ABSTRACT

This study evaluates the potential impact of increased biofuel production on the hydrology of a small watershed, Khlong Phlo, in the eastern part of Thailand. The water footprint of biofuel energy was estimated for three crops in order to identify the most water-efficient crop. The Soil and Water Assessment Tool (SWAT) model was used to evaluate the impact of land use change (LUC) caused by the expansion of biofuel crops on the components of water balance and water quality in the studied watershed. Several LUC scenarios consisting of oil palm (biodiesel), cassava and sugarcane (bio-ethanol) expansion were evaluated. The water footprint results indicated that cassava is more water-efficient than the other two crops considered. Simulation results revealed that although oil palm expansion would have negligible alteration in evapotranspiration (0.5 to 1.6%) and water yield (−0.5 to −1.1%), there would be an increased nitrate loading (1.3 to 51.7%) to the surface water. On the contrary, expansion of cassava and sugarcane would decrease evapotranspiration (0.8 to 11.8%) and increase water yield (1.6 to 18.0%), which would lead to increased sediment (10.9 to 91.5%), nitrate (1.9 to 44.5%) and total phosphorus (15.0 to 165.0%) loading to surface water. Based on the results, it can be concluded that land use change for biodiesel production would affect water quality, while both the water balance components and water quality would be affected by the expansion of bio-ethanol crops. Overall, the study indicates that biofuel production would have a negative impact on the water quality of the studied watershed. Further research at large scale (e.g. basin level) and on the economic aspect is recommended, in order to contribute to developing suitable land use and energy policies.

1. Introduction

Biofuels are fuels used for transport and are derived from biomass (Dufey, 2006). There are two types of liquid biofuel for transport: bio-ethanol and biodiesel. Starchy crops, sugar crops and cellulosic material are used to produce bio-ethanol, whereas biodiesel is produced from oil crops. Many countries (like the USA, Brazil, China, India, Thailand, Malaysia etc.) are promoting biofuel so as to cut down fossil fuel consumption, to decrease oil import, to reduce greenhouse gas emission, and to reduce the poverty level of rural communities (Dufey, 2006; De Fraiture et al., 2008). The global biofuel production, which was around 57 billion liters at the end of 2007, is projected to increase almost threefold by 2017 (OECD-FAO, 2008). Under conditions of increased biofuel production, land use is likely to incur significant changes as large amounts of land would be required for plantations (UNEP, 2008). For example, around 43 and 38% of the present cropland in the United States and Europe (respectively) will be needed to substitute just 10% of petrol and diesel fuel (IEA, 2005). Similarly De Fraiture et al. (2008) estimated that by 2030, biofuel would need 30 million additional hectares of cropped area globally to share 7.5% of the total global gasoline demand (1,747 billion liters). It is very likely that the expansion of biofuel crops (oil palm, sugarcane, soybeans, etc.) will replace native rainforests and wetlands (Muller et al., 2007) due to shortage of land to expand agriculture. In Asian countries like Malaysia and Indonesia, the cultivation of oil palm for biofuel production has already replaced a large part of the forest land cover (Meijerink et al., 2008).

The Thai government has plans to increase the share of renewable energy in the total energy consumption from 0.5% in 2002 to 20.3% (4.1% from biofuel) by 2022 (Preechajarn and Prasertsri, 2010). In Thailand, biofuel is projected to replace 4,928 million liters of fossil fuel annually by the year 2022 (Prasertsri and Kunasirirat, 2009). To meet increasing biodiesel demands, the “Committee on Biofuel Development and Promotion” targets to increase oil palm coverage by 0.4 million ha by 2012 (APEC, 2008) through orchard replacement which is already happening in the northern, north-eastern, eastern and southern regions of Thailand (Prasertsri and Kunasirirat, 2009). There are also government plans of expanding

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the oil palm land cover to 1.6 million ha by 2023 (Siriwardhana et al., 2009). In 2008 land coverage for oil palm, cassava and sugarcane was 0.5, 1.1 and 1.2 million ha, respectively (OAE, 2008). At present, all the feedstock left after satisfying exports and the domestic demand is used for biofuel production. A tonne of fresh bunch palm approximately yields 170 kg of crude palm oil and 50 kg of palm kernel. In 2008, about 25% of crude palm oil was utilized for biodiesel production while the rest was used for human consumption and others (including domestic use and export) (Silalertruksa and Gheewala, 2011). Both the fresh roots and dried chip of cassava can be used for bio-ethanol production. In 2008, about 37% of cassava was locally utilized in various industries including bio-ethanol (7%), starch production (21%) and cassava chips/strips and pillets (9%), while the remaining cassava was exported (DEDE, 2009). Molasses, a viscous by-product of the sugar-milling, currently plays an important role in the major feedstock for producing ethanol in Thailand. A tonne of sugarcane approximately yields 104 kg of sugar and 46 kg of molasses (OCSB, 2011). In 2008, about 78% of molasses was locally utilized in various industries including bio-ethanol (37%), animal feed and Monosodium glutamate (MSG) production (11%) and distilleries (30%), while the remaining molasses was exported (DEDE, 2009). Cane juice of sugarcane is one of the most crucial products for bio-ethanol production, but as of 2010 less than 1% of the sugarcane is used for bio-ethanol production and the rest for producing sugar (USDA, 2011). Nevertheless it is projected that of the total sugarcane yield 27% will be utilized for domestic sugar production, 44% for export and 59% for bio-ethanol (DEDE, 2009). With the current diversion of 3 million tonnes of cassava and 2 million tonnes of molasses annually, the bio-ethanol production potential of Thailand is roughly 3×10^6 L/d. The projection of 9×10^6 L/d of bio-ethanol by 2022 implies that in order to maintain domestic demand and export unaffected, Thailand needs either to intensify agriculture or expand the land for feedstock production. Thus, to meet the demand for biofuel, there would need to be considerable land use changes, which would add stress to the already limited water resources (Hoogeveen et al., 2009).

Land use change for increased biofuel production can have considerable impact on water resources and the aquatic environment. Replacing existing crops with biofuel crops can influence effective rainfall apart from altering the soil and climate due to a change in evapotranspiration and interception, which can have significant implications for surface runoff and groundwater recharge (Stephens et al., 2001). Large scale, intensive biofuel crop production requires a substantial amount of fertilizers (NRC, 2007; Evans and Cohen, 2009). Excessive use of agricultural fertilizers and their transport to water bodies through leaching and surface runoff can cause environmental problems like eutrophication and increase the level of nitrate and nitrite, which make water resources unusable for other purposes. Land use change due to biofuel production may have a significant impact on water quality (Hill et al., 2006; NRC, 2007; FAO, 2008; Schilling et al., 2008; Cruse and Herndl, 2009; Evans and Cohen, 2009; Gopalakrishna et al., 2009; Thomas et al., 2009; Twomey et al., 2009; Blanco-Canqui, 2010; De La Torre Ugarte et al., 2010; Delucchi, 2010). It is likely that, with increasing biofuel production, there will be severe impacts on hydrological processes and water cycle dynamics, but the quantification of these changes are complex (Uhlenbrook, 2007; IWMI, 2009; Meijerink et al., 2008; Engel et al., 2010).These impacts of various biofuel crop-production systems will be a function of feedstock of choice, watershed management and soil and climate conditions (Engel et al., 2010). Biofuel production can be sustainable if it has minimal hydrologic and water quality implication hence, there is a need of scientific assessment of regional feedstock production impacts on water resources and water quality which is recommended by many researchers (Shannon et al., 2008; Gopalakrishna et al., 2009; Engel et al., 2010).

In order to reduce the impact on water resources, it is necessary to find the most water-efficient crop to produce biofuel. The concept of water footprint has been in use only recently by researchers for biofuel production (Gerbens-Leenes et al., 2009; Yang et al., 2009; Mekonnen and Hoekstra, 2011). The water footprint is defined as "the total annual volume of fresh water required to produce the goods at the place of origin" (Hoekstra and Chapagain, 2007) and consists of three components: green, blue and grey water footprints. Green water footprint is rainwater that evaporated during production, mainly during crop growth; blue water footprint is irrigated surface water and groundwater which evaporates during crop growth; and grey water footprint is the amount of water needed to dilute pollutants discharged into the natural water system to the extent that the quality of the ambient water remains above agreed water quality standards (Gerbens-Leenes et al., 2009). The water footprint of biofuel energy depends upon the crop being cultivated, the yield of the crop, climatic conditions at the location of its production, and agricultural practices (Gerbens-Leenes et al., 2009; Yang et al., 2009).

Computer simulation models can be an effective tool to quantify the effects of biofuel crop production on hydrology and water quality at various spatial scales ranging from individual fields to watersheds and large river basins, and temporal scales ranging from individual storm events to annual and decades (Engel et al., 2010). The Soil and Water Assessment Tool (SWAT) is one of the models that has been extensively used to evaluate impacts of various land use, management and climate conditions on hydrologic and water quality response of agricultural and mixed land use watersheds (Borah and Bera, 2003). Love and Nejadhashemi (2011) applied the SWAT model to examine the possible long term water quality implication of large-scale biofuel crop expansion in agricultural watersheds of Michigan. Schilling et al. (2008) used SWAT to evaluate the potential impacts on the water balance components and water quality due to biofuel crop expansion in an agricultural watershed of west-central Iowa. Zhai et al. (2010) also applied SWAT to simulate the hydrologic and water quality impacts of increased biofuel production in the Upper Mississipi River Basin.

In this paper, the potential impact of land use change due to biofuel production on the hydrology and water quality of a small watershed, Khlong Phlo, in the eastern part of Thailand, was evaluated. The study first estimates the water footprint of biofuel energy from three main crops: oil palm (*Elaeis guineensis Jacq.*), cassava (*Manihot esculenta Crantz*) and sugarcane (*Saccharum officinarum*) grown in the study area, and then analyzes the impact of land use change due to an expansion of the biofuel crops on annual and monthly water balance components, and on the water quality in the Khlong Phlo watershed using the SWAT model.

2. Materials and methods

2.1. Study area

Khlong Phlo is a subbasin of the Khlong Prasae basin located in the eastern part of Thailand (Fig. 1). The studied watershed lies within 12°57′–13°10′N and 101°35′–101°45′E and encompasses a total land area of 202.8 km^2 above the stream gauge station Z.18 operated by the Royal Irrigation Department (RID). The elevation of the watershed ranges from 13 m above mean sea level at its lowest point to 723 m at its highest point. The annual mean temperature ranges from 27 to 31 °C and the relative humidity ranges from 69 to 83%. The watershed receives an average annual rainfall of 1,734 mm, of which 85% falls from May to October. The average discharge of the Phlo Stream at Z.18 is 6.7 m^3/s in May–October (wet season) and 0.7 m^3/s in November–April (dry season).

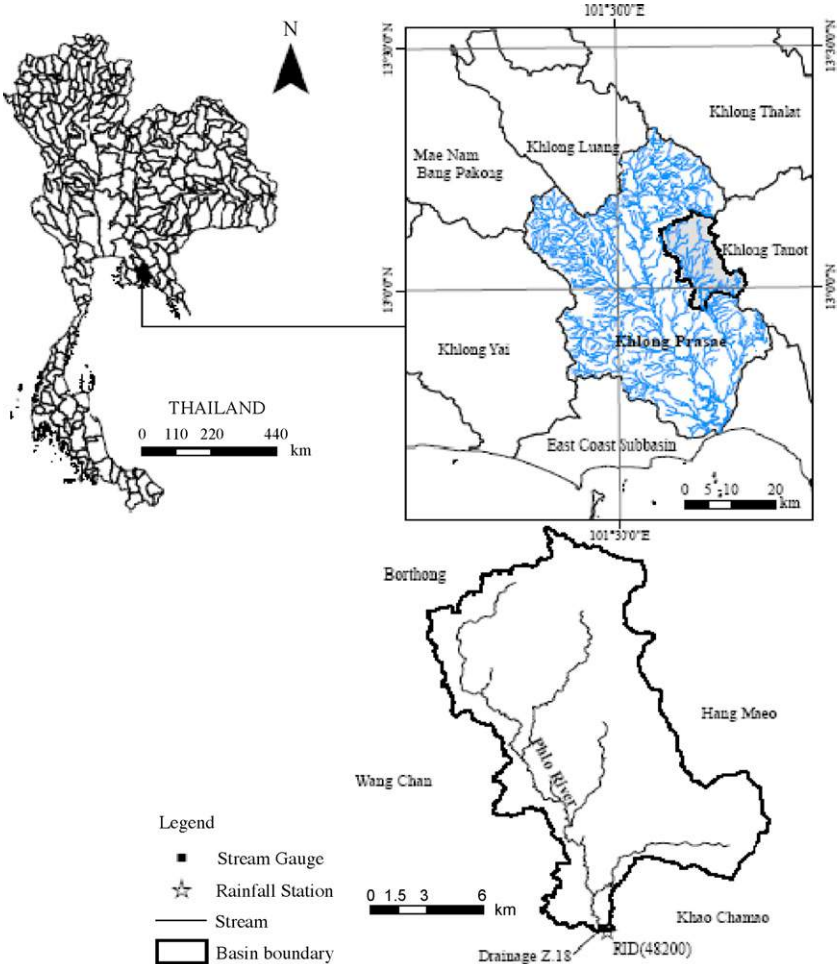

Fig. 1. Location of the Khlong Phlo watershed, stream gauge and rainfall stations.

The average annual observed runoff (573.7 mm) is approximately 33% of the rainfall and almost 90% of the discharge occurs in the wet season. Fig. 2 shows the flow duration curve (FDC) for the Khlong Phlo river, which indicates that the river is non-perennial, has large variability in flow and most of the discharge is contributed by the wet seasons flow. The river runs dry for several months in some years. This is reflected in the FDC showing that about 27% of the time flow is zero. The watershed yields, on average (9-year average), 11,488 tonnes of sediment annually at Z.18. The average annual sediment concentration is nearly 126 mg/L. Agricultural land is the dominant land cover in the watershed, which is nearly 66% of the total area (Table 1). The land used for biofuel crops comprises roughly 10% of the total agricultural land in the area. Soils in this watershed are predominantly sandy clay loam and sandy loam. The seven major soil types in the study area are presented in Table 2.

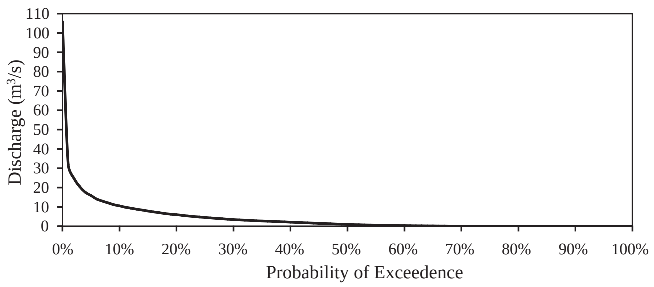

Fig. 2. Flow duration curve for the Khlong Phlo River at Z.18.

2.2. Estimation of water footprints

In this study, the water footprint is calculated at farm gate level only because water use in the life cycles of products is dominated by the agricultural production stage (Gerbens-Leenes et al., 2009). The

Table 1
Land use in the Khlong Phlo watershed.

Land use	Area (km^2)	Percent
Agriculture	132.9	65.5
Rubber	85.1	64.1
Orchard	28.0	21.0
Cassava[a]	9.9	7.4
Cashew nut	4.8	3.6
Sugarcane[a]	2.1	1.6
Rice	1.8	1.4
Oil palm[a]	1.1	0.8
Forest	66.4	32.8
Range	1.8	0.9
Urban	0.8	0.4
Water bodies	0.9	0.4

Source: Land Development Department, Thailand (2010).

[a] Crops used for biofuel production (area = 13.1 km^2, roughly 10% of agricultural land).

Table 2
Soil series and hydraulic properties in the Khlong Phlo watershed.

Soil Series	Depth of soil layer (mm)	Textural name	Composition (%)			Organic carbon content (%)	Bulk density (g/cm^3)	Available water capacity (mm/mm)	Saturated hydraulic conductivity (mm/h)
			Sand	Silt	Clay				
Nong Khla	0–130	Sandy clay loam	54	25	22	2.3	1.4	0.1	18.5
	130–300	Clay	39	20	42	1.2	1.4	0.1	1.4
	300–490	Clay	14	8	78	0.7	1.2	0.1	0.9
	490–1000	Clay	16	8	76	0.4	1.2	0.1	0.7
Lamphu La	0–110	Silty loam	11	68	21	3.2	1.1	0.2	30.6
	110–280	Silty clay loam	8	60	33	2.7	1.2	0.2	13.3
	280–420	Silty clay	8	52	41	1.2	1.3	0.2	4.0
	420–1000	Clay	6	40	55	1.8	1.2	0.1	3.6
Khlong Teng	0–120	Silty loam	23	57	21	2.2	1.3	0.2	18.0
	120–210	Clay loam	21	51	29	1.4	1.4	0.2	7.2
	210–370	Clay	21	39	41	1.5	1.3	0.1	3.2
	370–1000	Clay	8	36	56	0.9	1.2	0.1	1.9
Tha Sae	0–210	Sandy loam	81	7	13	0.8	1.6	0.1	44.9
	210–500	Sandy loam	58	23	19	0.2	1.6	0.1	15.0
	500–670	Sandy clay loam	55	20	25	0.1	1.6	0.1	7.6
	670–1000	Sandy clay loam	54	18	29	0.1	1.6	0.1	4.8
Huai Pong	0–170	Sandy loam	63	25	13	1.6	1.5	0.1	39.3
	170–400	Sandy clay loam	51	24	26	0.6	1.5	0.1	7.6
	400–680	Sandy clay loam	49	22	30	0.4	1.5	0.1	4.5
	680–750+	Clay	40	15	46	0.3	1.5	0.1	0.5
Phang-nga	0–140	Sandy loam	56	37	7	1.0	1.5	0.1	49.4
	140–290	Sandy loam	66	16	19	1.9	1.5	0.1	24.5
	290–670	Sandy clay	55	10	36	0.7	1.5	0.1	2.2
	670–1100	Sandy clay	53	7	40	0.5	1.5	0.1	1.1
Orthic acrisols	0–300	Sandy clay loam	49	27	24	1.0	1.5	0.1	8.7
	300–1000	Clay loam	40	24	36	0.4	1.5	0.1	1.9
Ferric acrisols	0–300	Sandy loam	78	12	10	0.6	1.6	0.1	51.3
	300–1000	Sandy clay loam	65	12	23	0.3	1.6	0.1	10.0

water footprint of biofuel energy was estimated following several steps, as presented below.

2.2.1. *Water footprint of crops (WF_C)*

The water footprint of the crops was calculated based on the method suggested by Chapagain and Orr (2009), where the water footprint of the crop WFc (m^3/t) is the proportion of the amount of water used to produce the crop Wc (m^3/ha), to the crop yield, Y (t/ha):

$$\mathrm{WFc} = \frac{\mathrm{Wc}}{Y} \quad (1)$$

Here, the crop yield (Y) represents a mass of fresh, ready-for-process fruit bunches, cane stalks, and cassava roots per ha, delivered at the farm gate, for oil palm, sugarcane and cassava crops respectively.

The amount of water used for crop production (Wc) is composed of two components:

$$\mathrm{Wc} = W_{\mathrm{evaporative}} + W_{\mathrm{non\text{-}evaporative}} \quad (2)$$

where $W_{\mathrm{evaporative}}$ is the volume of water evaporated and $W_{\mathrm{non\text{-}evaporative}}$ is the volume of water unavailable for further use as a result of pollution. These components are:

$$W_{\mathrm{evaporative}} = \text{Green water use} + \text{Blue water use} \quad (3)$$

$$W_{\mathrm{non\text{-}evaporative}} = \text{Grey water use} \quad (4)$$

Therefore, the WFc is a sum of evaporative and non-evaporative components:

$$\mathrm{WFc} = \frac{\text{Green water use}}{Y} + \frac{\text{Blue water use}}{Y} + \frac{\text{Grey water use}}{Y} \quad (5)$$

$W_{\mathrm{evaporative}}$ is classified into green and blue water use based on crop evapotranspiration ($\mathrm{ET_c}$) and effective rainfall (P_{eff}). The $\mathrm{ET_c}$ was calculated using the crop coefficient (k_c) for the respective growth period and reference crop evaporation ($\mathrm{ET_o}$) at that particular location and time using the equation:

$$\mathrm{ET_c} = k_c \times \mathrm{ET_o} \quad (6)$$

$\mathrm{ET_o}$ was estimated using the Penman-Monteith equation as recommended by FAO (Allen et al., 1998). The $\mathrm{ET_c}$ is based on the crop water requirement assuming optimal conditions and not including a dynamic soil water balance.

The green water use (m^3/ha) over the length of the growing period was calculated as the sum of daily volumes of rainwater evapotranspiration. This green water use is the minimum value of either P_{eff} or $\mathrm{ET_c}$:

$$\text{Green water use} = \min(\mathrm{ET_c}, P_{\mathrm{eff}}) \quad (7)$$

The blue water use (m^3/ha) over the length of the growing period was calculated as the sum of daily volumes of irrigation-water evapotranspiration. This blue water use is the minimum of either the irrigation requirement (I) or the effective irrigation supply (I_{eff}):

$$\text{Blue water use} = \min(I, I_{\mathrm{eff}}) \quad (8)$$

Here, the irrigation requirement (I) is calculated as:

$$I = \mathrm{ET_c} - P_{\mathrm{eff}} \quad (9)$$

I_{eff} is irrigation water stored as soil moisture and available for crop evaporation. In cases where the entire $\mathrm{ET_c}$ is met by P_{eff}, blue water use is zero.

Table 3
Nitrogen application rate, biofuel crop yield and conversion rate.

Crop	Nitrogen application rate (kg/ha)	Average yield[a] (t/ha)	Biofuel produced	Conversion rate (L/t)
Oil palm	105[b]	12.4 (fresh fruits)	Biodiesel	221[d]
Cassava	100[c]	23.6 (fresh roots)	Bio-ethanol	180[e]
Sugarcane	75[c]	62.1 (fresh cane)	Bio-ethanol	70[e]

[a] Office of Agricultural Economics (personal communication).
[b] http://www.fao.org/ag/agl/fertistat/fst.fubc.en.asap (accessed September 06, 2009).
[c] Department of Agriculture (personal communication).
[d] Gonsalves (2006).
[e] Department of Alternative Energy Development and Efficiency, 2006 as cited in Martchamadol, 2007.

The grey water use (m^3/ha) is the ratio of the weight of a given pollutant actually released into the water system due to crop production (Pr) to the permissible limit of that pollutant (Pl):

$$\text{Grey water use} = \frac{Pr}{\text{Pl}} \tag{10}$$

The grey water use was calculated based on the maximum acceptable drinking water quality standard for N-Nitrate (10 mg/L) and leached nitrogen to surface water. The calculation considers only nitrogen as a pollutant due to unavailability of standards for potassium and phosphorus in Thailand's water quality standards as well as in other standards like US-EPA and WHO. Modern agrochemicals used have relatively short half-lives, and are less hazardous to the environment because they deactivate faster. Based on this assumption, agrochemicals were also not included while calculating the grey water. The grey water use calculation by previous researchers (Chapagain et al., 2006; Van Oel et al., 2008; Scholten, 2009) assumed that only 10% of the total nitrogen applied reached surface water. This rate was based on two assumptions: (i) the plant removes 60% of the applied nitrogen; and (ii) in the long run, there will be a steady state balance at the root zone. On average, cassava removes nearly 55% (Howler, 1991), sugarcane 57% (Howler, 1991), and oil palm 70% (Ng, 1977) of applied nitrogen. This clearly justifies the assumption that 60% of the applied nitrogen is removed by plants. The manual on Water Footprint (Hoekstra et al., 2009) also suggests that one can assume a 10% leaching fraction of nitrogen fertilizers if no observed data is available. Such was the case in this study. Apart from the 10%, a sensitivity analysis approach was also used for the grey water use calculation. The crop yield used in the calculation is the average yield of three production years (2006–2008) (Table 3).

2.2.2. The water footprint of biofuel (WF_B)

The water footprint of biofuels (L of H_2O per L of biofuel) was calculated by dividing the water footprint of crop (m^3/t) times 1000 by biofuel conversion rate (L/t). The biofuel conversion rate of different crops used in the study is presented in Table 3.

2.2.3. The water footprint of biofuel energy (WF_{BE})

The water footprint of biofuel energy (m^3 per GJ of biofuel energy) was estimated by dividing the water footprint of the biofuel (L of H_2O per L of biofuel) times 1000 by the biofuel energy content (kJ/L). The energy per liter of biofuel is calculated by multiplying the Higher Heating Value (HHV) of the biofuel (kJ/g) and the density of the biofuel (kg/L) by 1000. The Higher Heating Value (HHV) and density of the biofuel were adopted based on available literature (Table 4).

Table 4
Higher heating value (HHV) and density of biofuel.

Biofuel	Higher heating value (HHV)[a] (kJ/g)	Density[b] (kg/L)
Biodiesel	37.7	0.84
Bio-ethanol	29.7	0.79

[a] Penning de Vries et al. (1989) and Verkerk et al. (1986).
[b] http://www.dft.gov.uk/pgr/roads/environment/research (accessed February 5, 2010).

2.3. The evaluation of the impact on water balance components and water quality

The Soil and Water Assessment Tool (SWAT) model was used to evaluate the impact on the water balance components and water quality in the studied watershed due to land use change for biofuel production. The SWAT model was used because it is physically based and simulates actual processes, it originated from agricultural models, and because of the degree of software support available to users. In addition SWAT has been validated in many regions of the world (Jha et al., 2007) and used for predicting hydrological and water quality impacts of biofuel crop (Gassman et al., 2004; Schilling et al., 2008; Baskaran et al., 2010; Ng et al., 2010; Zhai et al., 2010; Love and Nejadhashemi, 2011).

2.3.1. The SWAT model

SWAT is a river basin or watershed scale hydrologic and water quality model developed by Arnold et al. (1993). The model evaluates the effect of land use management on water, sedimentation, and agricultural chemical yields in large complex watersheds which are heterogeneous in land use, soil and management conditions over a long period of time. It is a semi-distributed, physically based, computationally efficient, and continuous time model. It is also capable of simulating a high level of spatial detail. Readers are referred to Neitsch et al. (2005) for the theoretical background of the SWAT model and other details.

2.3.2. Model inputs

Climatic data on daily temperature, wind speed, humidity and sunshine hours from 1984 to 2006 were obtained from the Thai Meteorological Department (TMD), while the daily rainfall data (for 1984 to 2006) were obtained from both the TMD and the Royal Irrigation Department (RID). Daily discharge (for 1984–2006) and sediment yield (for 1997–2005) data from the gauging station at Z.18 in the study area and a drainage map of the Phlo river watershed were obtained from the RID. A 30 m resolution Digital Elevation Model (DEM) of the study watershed was downloaded from http://www.gdem.aster.ersdac.or.jp.

A land use map of the year 2006 with a scale of 1:25,000 for the studied watershed was obtained from the Land Development Department (LDD). The land cover comprises 65.5% agriculture crops predominantly rubber and orchard and 32.8% forests (Table 1). For hydrological response unit (HRU) definition purposes, the land use map from the LDD was reclassified based on SWAT land use classification. The physical properties for cassava are based on the studies by Connor et al. (1981); Oka et al. (1989); Fageria and Baligar (1991); Morgan (1995) and El-Sharkawy (2004). The leaf area index of rubber is taken from Webster and Baulkwill (1989). SWAT default values were assigned for the curve number (CN) of all land uses in the study area and for the remaining physical properties of other land uses because the local values for these parameters were not available.

The soil distribution map (1:100,000) and the physical soil properties in the Khlong Phlo watershed were obtained from the Land Development Department (LDD). Soil series and hydraulic properties in the studied watershed are presented in Table 2. The LDD has defined the soil types of the hilly topography as slope complex (SC). The properties of SC soil were not studied by LDD. Hence, the properties for SC soil were extracted from the Harmonized World Soil Database developed by the Land Use Change and Agriculture Program of IIASA (LUC) and the Food and Agriculture Organization of the United Nations (FAO). The source of this database is the website http://www.iiasa.ac.at/Research/LUC/External-World-soil-database/HTML/. The hydraulic properties of this soil type were calculated using the Soil-Plant-Air-Water Model (SPAW model, Saxton and Willey, 2005) based on the soil textures, compositions and organic matter. The soil database for the watershed was added to the "usersoil" database file and the soil map was reclassified based on the provided usersoil.

The fertilizer application rate was provided by the Department of Agriculture and obtained from www.fao.org/ag/agl/fertistat/fst.fubc.en.asap. Agricultural practices were obtained based on interviews with local farmers, literature, and common assumptions in order to provide appropriate inputs for the model. For fertilizer application, the elemental nitrogen and elemental phosphorus options in SWAT were used. For other management data, SWAT default values were used.

2.3.3. Model calibration and validation

For this study the ArcSWAT (version 2.3.3) model was used. The Khlong Phlo watershed SWAT model was calibrated and validated for a monthly streamflow and sediment load. The calibration period for the streamflow was 1986–1995 (10 years) and the validation period was 1996–2000 (5 years). The sediment yield was calibrated for 3 years (1997–1999) and validated for the year 2000. Due to unavailability of measured data, the model was only simulated for nitrogen and phosphorus. Prior to the calibration process, the baseflow was separated from the surface flow for the observed streamflow using an automated baseflow method developed by Arnold and Allen (1999). This method uses a recursive digital filter technique which filters surface runoff (high frequency signals) from baseflow (low frequency signals).

The model performance is evaluated using the percentage difference between the simulated and observed values of the mean and standard deviation of the variables considered over the simulated period, the coefficient of determination (R^2) and the Nash–Sutcliffe (NS) measure (Nash and Sutcliffe, 1970).

2.4. Land use change scenarios

The calibrated model was used to evaluate the impact of land use change on the water balance components, sediment yield, nitrogen and phosphorus loss in the Khlong Phlo watershed. The model was simulated for 23 years (1984–2006) for each scenario to evaluate the effects on the annual and monthly water balance components, while the model was simulated for 9 years (1997–2005) to assess the change on annual sediment, nitrogen and phosphorus loss. Table 5 presents the evaluated land use change scenarios, consisting of oil palm, cassava and sugarcane expansion. The land use of

Table 5
Details of the land use change scenarios in the Khlong Phlo watershed.

Scenario	Land use												Conversion
	Rubber		Forest		Orchard		Cassava		Sugarcane		Oil Palm		
	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	km^2	%	
Baseline (Existing)	85.1	42.0	66.4	32.7	32.8	16.2	9.9	4.9	2.1	1.0	1.12	0.6	
A. Oil Palm expansion													
A1	85.1	42.0	66.4	32.7	–	–	9.9	4.9	2.1	1.0	33.9	16.7	Orchard to oil palm
A2	–	–	66.4	32.7	32.8	16.2	9.9	4.9	2.1	1.0	86.2	42.5	Rubber to oil palm
A3	–	–	66.4	32.7	–	–	9.9	4.9	2.1	1.0	119.0	58.7	Orchard and Rubber to oil palm
A4	85.1	42.0	–	–	32.8	16.2	9.9	4.9	2.1	1.0	67.5	33.3	Forest to oil palm
A5	–	–	–	–	–	–	9.9	4.9	2.1	1.0	185.4	91.4	Orchard, Rubber and Forest to oil palm
B. Cassava expansion													
B1	85.1	42.0	66.4	32.7	–	–	42.7	21.1	2.1	1.0	1.1	0.6	Orchard to cassava
B2	–	–	66.4	32.7	32.8	16.2	95.0	46.9	2.1	1.0	1.1	0.6	Rubber to cassava
B3	–	–	66.4	32.7	–	–	127.8	63.0	2.1	1.0	1.1	0.6	Orchard and Rubber to cassava
B4	85.1	42.0	–	–	32.8	16.2	76.2	37.6	2.1	1.0	1.1	0.6	Forest to cassava
B5	–	–	–	–	–	–	194.2	95.8	2.1	1.0	1.1	0.6	Orchard, Rubber and Forest to cassava
C. Sugarcane expansion													
C1	85.1	42.0	66.4	32.7	–	–	9.9	4.9	34.9	17.2	1.1	0.6	Orchard to sugarcane
C2	–	–	66.4	32.7	32.8	16.2	9.9	4.9	87.2	43.0	1.1	0.6	Rubber to sugarcane
C3	–	–	66.4	32.7	–	–	9.9	4.9	120.0	59.2	1.1	0.6	Orchard and Rubber to sugarcane
C4	85.1	42.0	–	–	32.8	16.2	9.9	4.9	68.5	33.8	1.1	0.6	Forest to sugarcane
C5	–	–	–	–	–	–	9.9	4.9	186.4	91.9	1.1	0.6	Orchard, Rubber and Forest to sugarcane
D. Combined expansion													
D1	85.1	42.0	66.4	32.7	–	–	26.28	12.96	2.11	1.04	17.52	8.64	Orchard to oil palm and cassava
D2	–	–	66.4	32.7	32.8	16.2	52.44	25.86	2.11	1.04	43.68	21.54	Rubber to oil palm and cassava
D3	85.1	42.0	66.4	32.7	–	–	26.28	12.96	18.51	9.13	1.12	0.55	Orchard to cassava and sugarcane
D4	–	–	66.4	32.7	32.8	16.2	52.44	25.86	44.67	22.03	1.12	0.55	Rubber to cassava and sugarcane

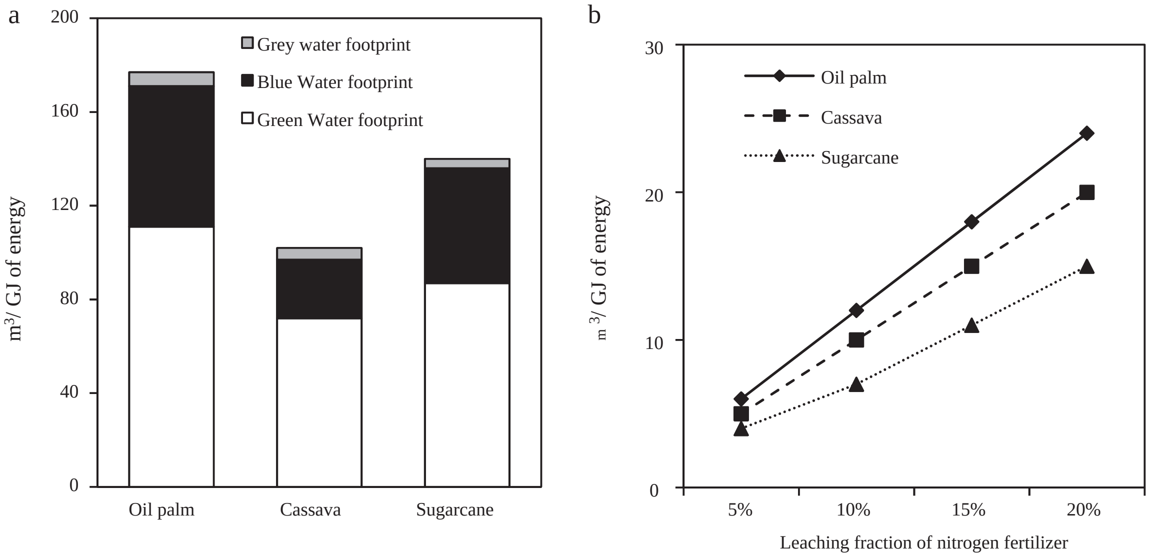

Fig. 3. Water footprint of biofuel energy, (a) green, blue and grey water footprints and (b) grey water footprint under different leaching fraction of nitrogen fertilizer.

year 2006 was used as the base case scenario (present) for the study. The present land use comprises 42% rubber, 32.7% forest, 16.2% orchard, 4.9% cassava, 1% sugarcane and 0.6% oil palm. Scenario A1, which refers to an increase in the oil palm area from less than 1 to nearly 17%, is motivated by the government plans to expand palm production through orchard conversion, developed by the Committee on Biofuel Development and Promotion (CBDP), jointly formed by the Ministry of Agriculture and Cooperatives and the Ministry of Energy, and by responses of palm ranchers to palm promotion, who have begun growing palm in new areas including the northern, northeastern, eastern and southern regions of Thailand by replacing old orchards (Prasertsri and Kunasirirat, 2009). Scenario A2 (rubber conversion and palm oil expansion) is justified on economic basis, since oil palm is a better alternative to rubber in areas where the two crops can be grown, because the internal rate of return for oil palm is a lot higher due to early harvesting as compared to rubber (Chirpanda et al., 2008). For environmental sustainability, the implications on water quantity and quality need to be evaluated if oil palm replaces rubber. Scenario A3 is the combined conversion of both orchard and rubber into oil palm which is investigated because if government encourages farmers to replace orchard and rubber for oil palm there is a possibility of this combined replacement in the study area where both orchard and rubber are grown.

The other alternative scenarios have been hypothesized. The forest replacement (Scenarios A4, B4, and C4) is investigated on the assumption that these scenarios may occur in the future for security of feedstock supply to meet an increased demand of biofuels and demand of the first generation crops for human consumption and animal feed. Expansion of cassava and sugarcane cultivation areas (Scenarios B1, B2, B3, C1, C2, and C3) are investigated with the hypothesis that the projected demand of 9×10^6 L/d of bio-ethanol by 2022 would be met by the expansion of cassava and sugarcane to keep domestic demands and exports unaffected, and for food security reasons replacement of orchard and rubber seems valid. Scenarios A5, B5 and C5 (conversion of orchard, rubber and forest together to oil palm, cassava or sugarcane) are assessed to investigate the impacts of extreme land use changes in the studied watershed. To analyze the potential impacts of combined expansions, D1, D2, D3 and D4 scenarios (Table 5) are simulated. It is assumed that the replacement would be distributed equally to biofuel crops. In these scenarios areas under orchard and rubber are converted as these changes are most likely to occur. Gradual conversion of rubber plantation area to oil palm, cassava and sugarcane is also investigated because land use conversions are likely to happen in a gradual fashion and assessment of change in water balance components and water quality would possibly allow identifying critical threshold areas of biofuel crops beyond which changes in water quantity and water quality becomes critical.

3. Results and discussion

3.1. *The water footprint of biofuel energy*

Fig. 3a shows the green, blue and grey water footprints (under 5% leaching fraction of nitrogen fertilizer) of biofuel energy production from three different crops. It requires 177, 103 and 140 m^3 of water to produce 1 GJ of biofuel energy from oil palm, cassava, and sugarcane, respectively, when nitrogen leaching is assumed to be 5%. In case of a maximum amount of the pollutant reaching surface water (20%), 1 GJ of energy requires a total of 200 m^3 of water for oil palm. Under the same conditions, cassava and sugarcane consume almost 120 and 150 m^3 of water, respectively, to produce one unit of bio-ethanol energy. Fig. 3b features the grey water footprint under four different leaching fractions of nitrogen fertilizer; it highlights that grey water varies with the fraction of applied nitrogen reaching surface water. The results reveal that to produce biofuel energy, the most water-efficient crop for the study area is cassava and the water footprint for bio-ethanol energy is less than for biodiesel which was also concluded by previous studies (Gerbens-Leenes et al., 2008; Mekonnen and Hoekstra, 2011). Biofuel production utilizing cassava as feedstock would have less impact on the water resources of the studied watershed as compared to sugarcane and oil palm.

In the United States of America, it requires 78 and 443 m^3 of evaporative water (lost through evapotranspiration, i.e. green and blue water) to produce 1 GJ of biofuel energy from maize and soybean respectively, while in Brazil 99 and 320 m^3 of evaporative water is used to produce 1 GJ of biofuel energy from sugarcane and soybean respectively (Gerbens-Leenes et al., 2008). The present study shows that producing 1 GJ of biofuel energy from cassava, sugarcane and oil palm requires 98, 136 and 171 m^3 of evaporative water, respectively. These results indicate that although the water footprint of bio-ethanol energy does not differ much from the largest bio-ethanol producers in the world, it requires less water to produce biodiesel energy in Thailand compared to Brazil and the

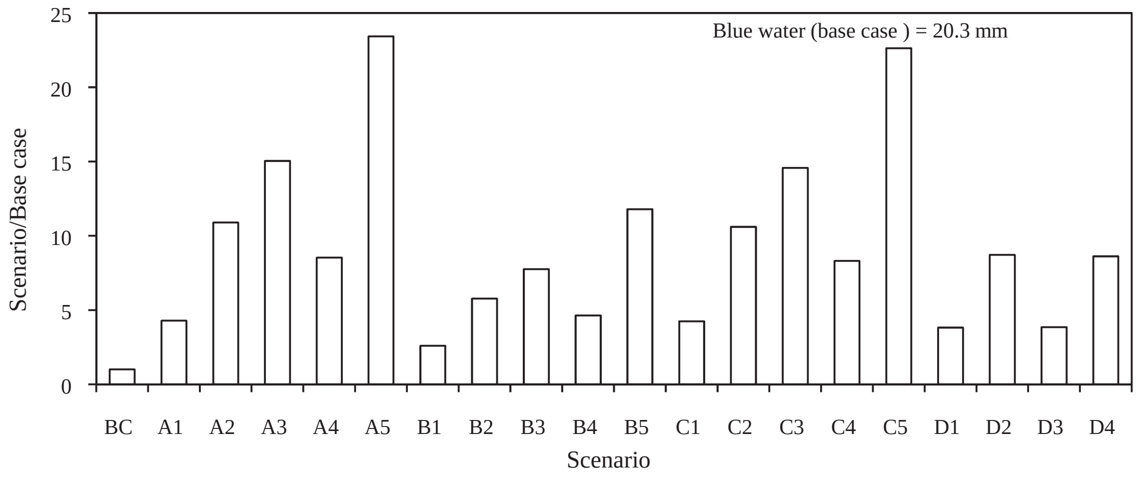

Fig. 4. Blue water requirement under different scenarios shown as ratio to the base case (BC).

USA. A study by Mekonnen and Hoekstra (2011) on the water footprints of several biofuel crops also concluded that oil palm is one of the most water efficient crops to produce biodiesel. However, the water requirements of biofuel production depend on feedstock used and on geographic and climatic variables, and such factors must be considered to determine water requirements and identify critical scenarios and mitigation strategies (Dominguez-Faus et al., 2009).

A study by Gerbens-Leenes et al. (2008) on the water footprint of biofuel energy reported 79 and 8 m^3 of green and blue water footprint, respectively, for cassava, and 64 and 55 m^3, respectively, for sugarcane to produce 1 GJ of bio-ethanol energy in Thailand. The total water lost through evapotranspiration (green and blue water footprint) for cassava is 10% less and for sugarcane it is 13% less as compared to the value calculated in this study area. This is due to a difference in crop water requirement and the yield used. The method used to calculate the crop water requirement is sensitive to the climatic data and to the cropping calendar. Climatic data and the planting season vary with location. For instance, in the present study area, the planting season for cassava is February, and Gerbens-Leenes et al. (2008) used April for their study. Gerbens-Leenes et al. (2008) used the climatic data from Chiang Mai for cassava and Nakhon Ratchasima for sugarcane. The yield used in the present study is based on the provincial average of three years (2006–2008) obtained from the Office of Agricultural Economics but Gerbens-Leenes et al. (2008) used the national average of 5 production years (1997–2001) obtained from FAO (sugarcane: 59.1 t/ha, and cassava: 15.9 t/ha). This simply suggests that the water footprint for biofuel energy is sensitive to the location; hence, the impact on the components of water balance would vary from place to place, thereby requiring localized studies.

3.2. Water footprint under various land use change scenarios

Fig. 4 shows the change in blue water requirement under different biofuel crop expansion scenarios as compared to the present land use in the study area. The results indicate that increase in cultivation area of biofuel crops would increase the blue water requirement which means increased irrigation withdrawals. This can have significant effect on the water resources availability in the studied watershed. For instance, under maximum biofuel crop expansion scenarios (A5, B5 and C5), where more than 90% of the land use is changed, the blue water requirement would increase by more than 20 times of the base case for oil palm (A5) and sugarcane (C5) and this amount of blue water is about 80% of the current total water yield (surface runoff plus baseflow) from the watershed. Such a large amount of irrigation withdrawals from surface water and groundwater sources for biofuel production would have serious implication on downstream water supply and environmental flow (McCornick et al., 2008).

As for the blue water requirement, the expansion of biofuel crops in the studied watershed would increase the grey water

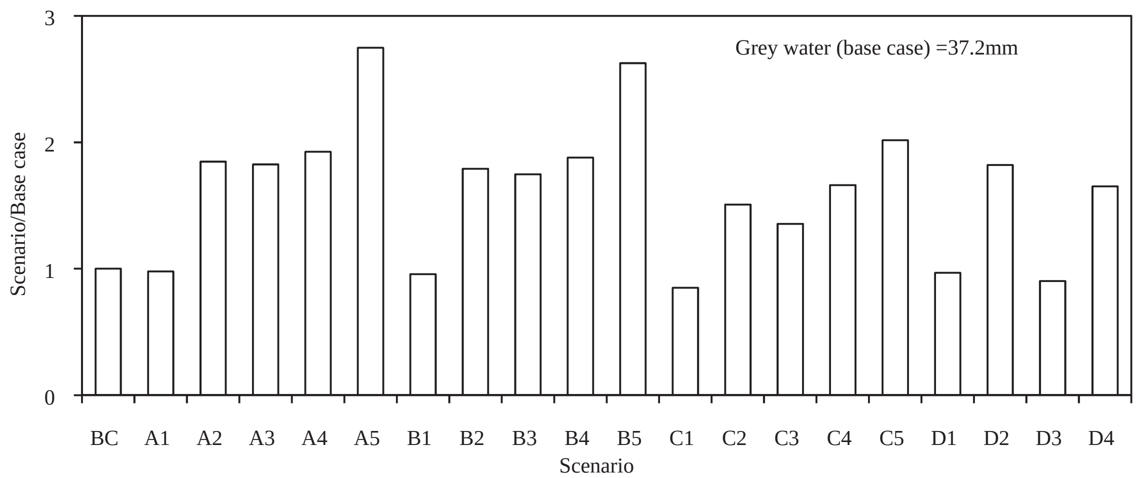

Fig. 5. Grey water requirement with a leaching fraction of 10% under different scenarios shown as ratio to the base case (BC).

Table 6
Observed and simulated annual average water balance components (1984–2006) for the Khlong Phlo watershed.

	Rainfall	Surface runoff	Baseflow	Evapotranspiration	Total water yield
Observed (mm)	1734.2	195.1	378.6[a]	–	573.7[b]
Simulated (mm)	–	206.7	389.8	835.6	596.5
Difference (%)	–	6.0	3.0	–	4.0

[a] Estimated based on automated baseflow method (Arnold and Allen, 1999).
[b] Total water yield = Surface runoff + baseflow.

requirement (Fig. 5), except for orchard replacement scenarios (A1, B1, C1, D1 and D3), due to change in fertilizer application rate. However, the decrease in grey water requirement for orchard replacement scenarios is negligible (less than 20% of the base case). The grey water requirement presented here is based on leaching 10% of applied nitrogen into surface water. With 90% of land cover dedicated to biofuel production the grey water use would increase more than two folds that of the current land use. This may be associated with replacement of forest area where no fertilizer is used as well as higher application of nitrogen in biofuel crops compared to rubber. Since grey water can be characterized indirectly as "water use for crop production" the increase in grey water requirement can reduce the availability of freshwater due to contamination of the water resources. This in turn adds more stress to water resources availability and hence decreased sustainability of biofuel production (Huffaker, 2010). To minimize the adverse impacts on water resources and water quality it is necessary to select biofuel crops which are not water intensive and use less fertilizer for their production (Dominguez-Faus et al., 2009). From the water resources perspective, the ideal land use change would be to replace orchard with biofuel crops.

3.3. Model calibration and validation

3.3.1. Calibrated parameters

The calibration process was performed following the procedure stated by Santhi et al. (2001) and in the SWAT Users Manual (Neitsch et al., 2002) to achieve a good fit between simulated and observed values. The calibration of flow and sediment was performed manually adjusting key hydrological and sediment related parameters. The parameters to be adjusted were selected based on suggestions by Santhi et al. (2001), Jha et al. (2007), and Schilling et al. (2008). The curve number (CN2), the soil evaporation compensation factor (ESCO), the available water capacity of the soil layers (SOL_AWC), the deep aquifer percolation fraction (RCHRG_DP), the groundwater revap coefficient (GW_REVAP) and the threshold depth of water in the shallow aquifer required for return flow to occur (GWQMN) were adjusted to improve the conformity between observed and simulated annual average surface runoff and baseflow and monthly streamflow. The linear (SPCON) and exponential (SPEXP) parameters for calculating the sediment transported in the channel sediment routing and the support practice factor or P factor of the Universal Soil Loss Equation (USLE) were adjusted to match the simulated and observed sediment yield. Adjustments of CN2, ESCO, GW_REVAP, SPECON and SPEXP have been found necessary by several SWAT studies, including Santhi et al. (2001), Jha et al. (2007), and Schilling et al. (2008).

3.3.2. Water balance components and streamflow

Table 6 compares observed and simulated annual water balance components of the study watershed. The annual average water yield of the watershed was predicted to be 596.5 mm, which consists of 206.7 mm surface runoff, and 389.8 mm baseflow. The total water yield and baseflow were modeled with a difference of less than 4%, and surface runoff with nearly 6% with respect to the observations, which reflect that the model was able to predict the total water yield and its components very well. The modeled baseflow proportion was 65% of the average annual flow which was almost the same as the estimated proportion of baseflow from the observed streamflow (66%). The simulated annual water balance of the watershed suggests that the total water yield (i.e., surface runoff plus baseflow) accounts for nearly 34% of the annual rainfall (1734.2 mm). The evapotranspiration and deep percolation represent 48 and 18% of the annual rainfall respectively.

Table 7
Monthly streamflow calibration and validation results.

	Calibration (1986–1995)			Validation (1996–2000)		
	Mean (mm)	SD (mm)	NS	Mean (mm)	SD (mm)	NS
Observed	59.5	86.9	0.8	46.5	52.8	0.6
Simulated	64.4	83.3		42.9	48.6	
Difference (%)	8.4	−4.2		−7.8	−8.0	

Fig. 6a compares the monthly simulated and observed streamflow at Z.18 for the calibration period. While the model represents the monthly streamflow well, flows were under-predicted for 1986, 1990 and 1994 and over-predicted for 1987 and 1992. The model calibration results in Table 7 indicate that the model can reasonably well simulate monthly streamflow with the difference in mean and standard deviation within 10% and with the Nash-Sutcliffe simulation efficiency (NS) greater than 0.5, which is an acceptable limit, as proposed by Santhi et al. (2001). Further, the coefficient of determination (R^2) of 0.8 and the proximity of the fitted regression line to the 1:1 line (Fig. 7a) shows that there is an acceptable linear relationship, and the model performance in predicting the stream flow is reasonable.

Fig. 6b shows that SWAT simulated the monthly flow for the validation period with reasonable accuracy. Further, the agreement between the modeled and observed flow of the Khlong Phlo watershed was confirmed by the mean and standard deviation difference being less than 10% and NS and R^2 were within acceptable limits, as proposed by Santhi et al. (2001) (Table 7 and Fig. 7b). Nevertheless, the streamflow was under-predicted for 2000 and time-to-peak for 1996, 1998, 1999 and 2000 were not represented very well. The linear relationship between the simulated and observed streamflow shown in Fig. 7b indicates that the streamflow for the validation period was not close to the ideal 1:1 correlation, as it was in the case of the calibration period. This may be due to a less accurate flow prediction for 1996 and 2000, as indicated by graphical comparison (Fig. 6b) which might be attributed to utilizing the land use of 2006 for both the calibration and validation processes against the measured data from 1984 to 2006. It would be ideal to use the actual land use data in the calibration and validation because land use may have changed over these 22 years and in reality the differences between observed and simulated data might be attributed to differences in the land use patterns.

Overall, the model was able to simulate annual water yield very well, and monthly streamflow with reasonable accuracy. These results indicate that the model can be extended to study the effects of various land use change scenarios on the components of water balance and streamflow.

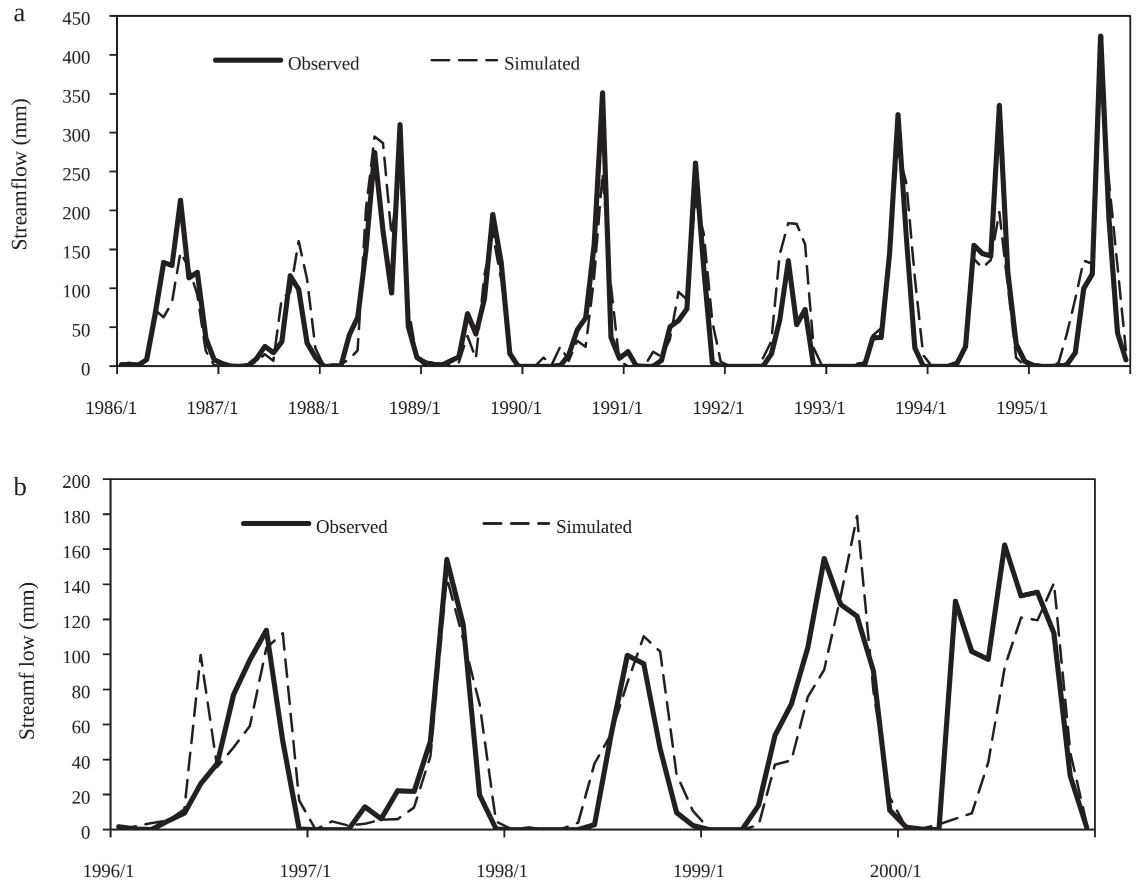

Fig. 6. Observed and simulated monthly streamflow for the Khlong Phlo watershed, (a) calibration period; (b) validation period.

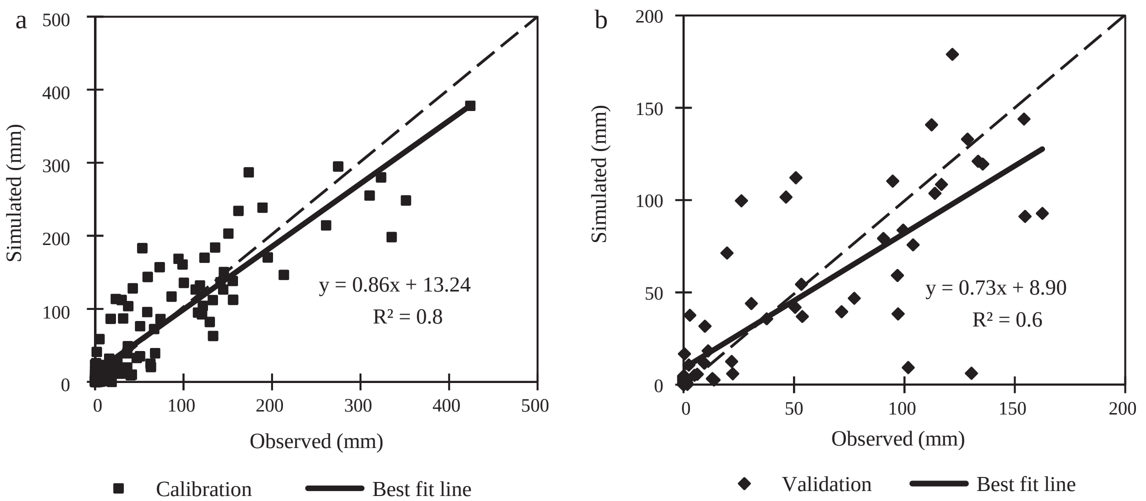

Fig. 7. Observed versus simulated monthly streamflow, (a) calibration; (b) validation.

3.3.3. *Sediment*

The model predicted an annual average sediment yield of 0.60 t/ha with an error of 5.1% relative to the observed amount (0.57 t/ha). Fig. 8 shows the observed and simulated monthly sediment yield for both calibration and validation periods. The SWAT model represented 1997 well, while the sediment yield was under-predicted for 1998 and 2000, and over-predicted for 1999.

Table 8 presents the results for both the calibration and validation of the monthly sediment yield. The results for the calibration were within acceptable limits set by Santhi et al. (2001), with a

Table 8
Monthly sediment yield calibration and validation results.

	Calibration Period (1997–1999)			Validation Period (2000)		
	Mean (t/ha)	SD (t/ha)	NS	Mean (t/ha)	SD (t/ha)	NS
Observed	0.059	0.095	0.7	0.096	0.089	0.4
Simulated	0.055	0.089		0.067	0.088	
Difference (%)	−7.0	−5.8		−30.0	−1.3	

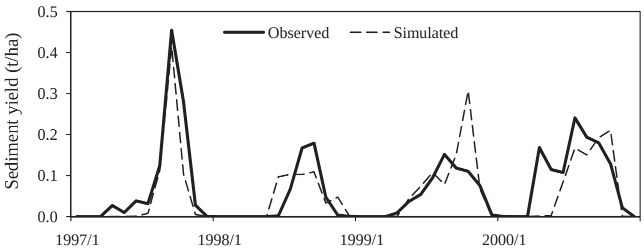

Fig. 8. Monthly sediment yield calibration (1997–1999) and validation (2000) for the Khlong Phlo watershed.

difference in mean and standard deviation between simulated and observed value of less than 10% and NS greater than 0.5. Also, the R^2 for the calibration was 0.7, as shown in Fig. 9a. This suggests that the model reasonably simulated the sediment yield for the calibration period. However, the model performance in predicting the sediment for the validation period was less accurate as compared to the calibration period, which is indicated by a mean difference of more than 10%, and NS of 0.4. Further, R^2 for validation was also less than the acceptable limit of 0.6 (Fig. 9b). Such differences in calibration and validation results for sediment yield have been reported in various previous studies. White and Chaubey (2005) reported that NS ranges from 0.2 to 0.8 and R^2 from 0.5 to 0.9 for monthly calibration and NS ranges from 0.3 to 0.9 and R^2 from 0.7 to 0.8 for monthly validation of sediment yield for three sub-watersheds of the Beaver Reservoir watershed in northeast Arkansas. Santhi et al. (2001) found NS greater than 0.5 and R^2 greater than 0.6 for monthly calibration but for validation NS was less than 0.5. The sediment component of SWAT uses the surface runoff volume and peak flow to compute the volume of sediment hence inaccuracies in simulating discharge may result in inaccurate estimates of sediment yields. The streamflow for 2000 was not represented very well (Fig. 6b). Further, the use of same land cover data for the calibration and validation processes might have caused such differences. In both cases, the fitted regression line was far from the ideal 1:1 line (Fig. 9a and b), which indicates that the SWAT model under-predicted the sediment load. The under-prediction of sediment load might be attributed to the number of rain gauges data used in the study. Chaplot et al. (2005) outlined that the use of a dense rain gauge network can significantly improve sediment predictions. In the present study only one rain gauge data was used because there were no other rain measurements available in the watershed. Moreover, this under-prediction can be due to an uncertainty in the

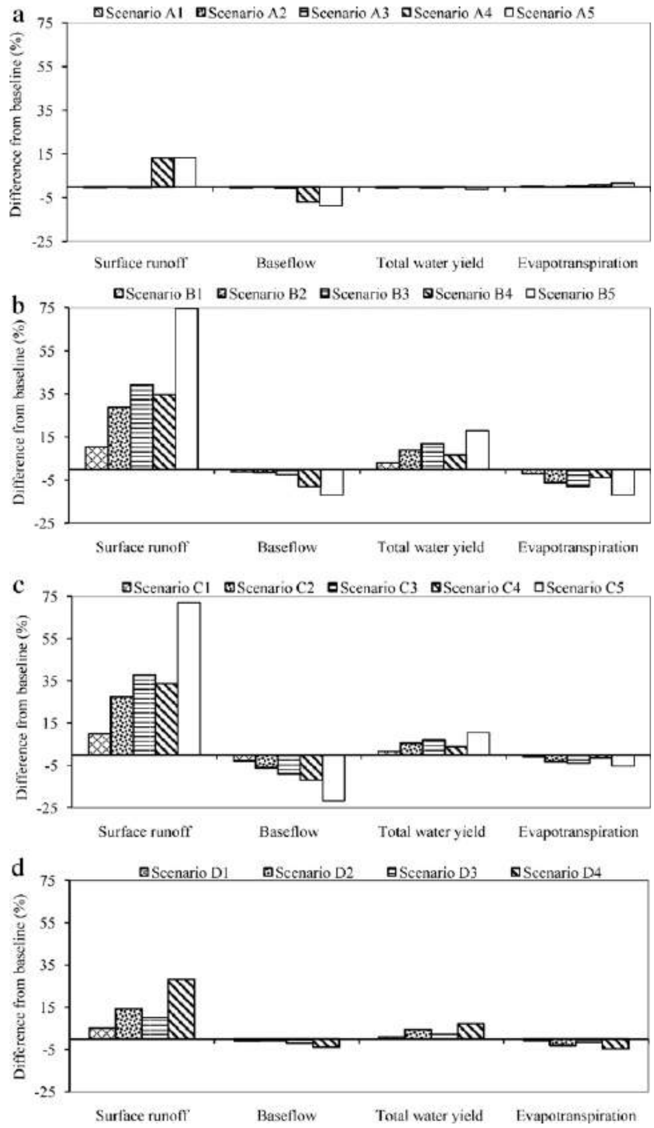

Fig. 10. Differences in annual average water balance components for different scenarios compared to the base case, (a) oil palm expansion; (b) cassava expansion; (c) sugarcane expansion; (d) combined expansion.

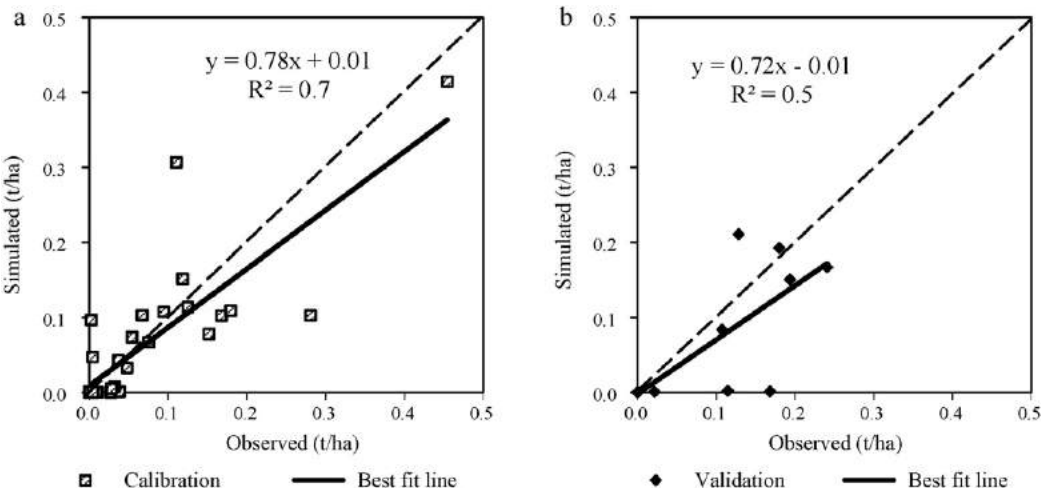

Fig. 9. Observed versus simulated monthly sediment yield, (a) calibration; (b) validation.

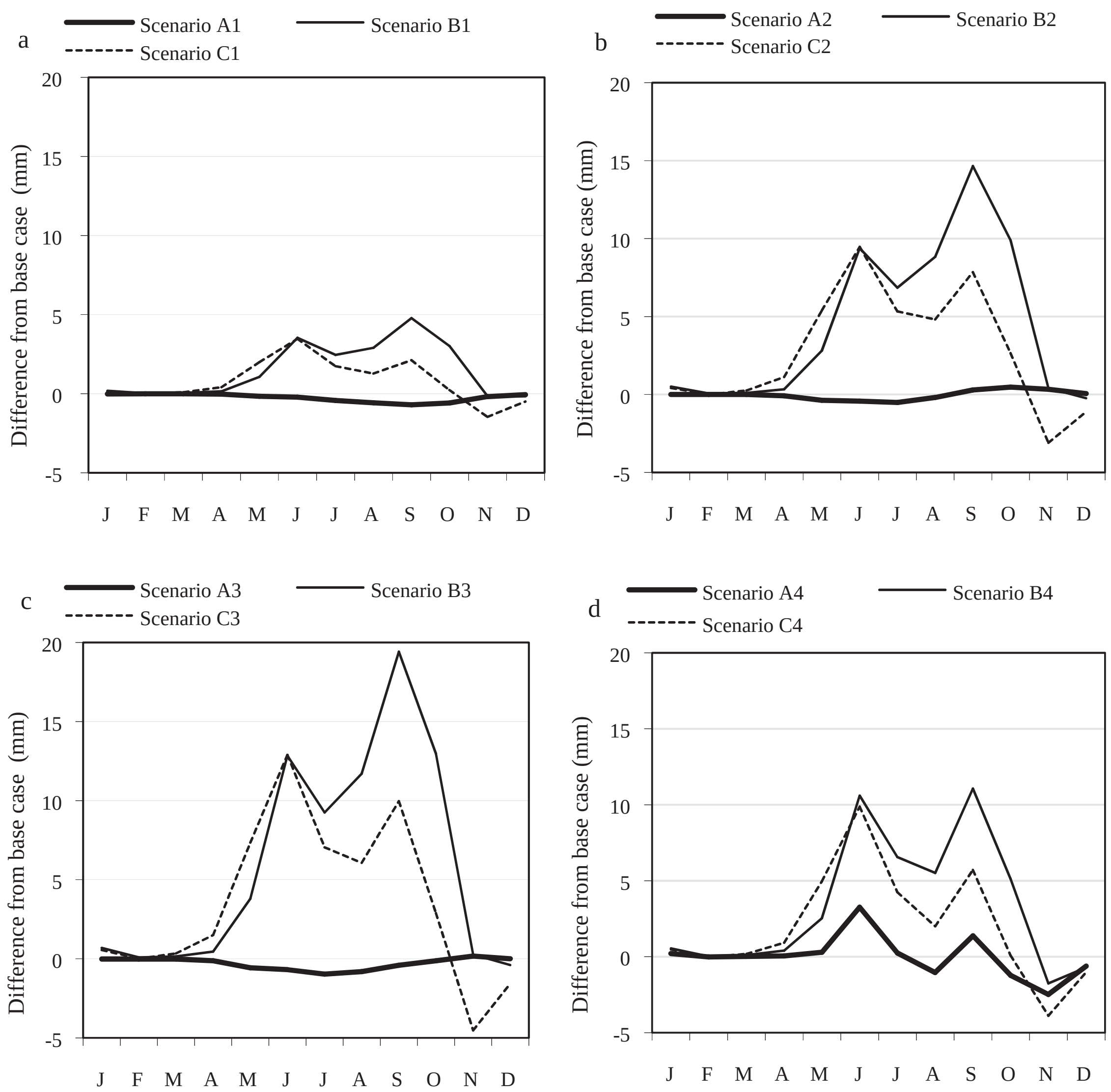

Fig. 11. Differences in average monthly water yield for different scenarios compared to the base case, (a) orchard area conversion; (b) rubber area conversion; (c) orchard and rubber area conversion; (d) forest area conversion.

soil erosion model used in SWAT. SWAT simulates erosion based on the Modified Universal Soil Loss Equation (MUSLE) (Williams, 1975) which is originally developed for estimates of annual soil loss from agricultural fields. Also, the topographic factor (LS) derived from DEM may not be accurate due to inaccuracies in DEM. Jackson et al. (1986) and Johnson et al. (1986) reported that MUSLE tends to over-predict sediment yields for small events and under-predict for large events. The studied watershed is located in a tropical climate with highly intensive rainfalls and heavy storms which have more potential to erode surface soil, but MUSLE does not account for such factors, which was also mentioned by Phomcha et al. (2011).

Overall, SWAT represented the annual average sediment yield accurately while the monthly time-series of observed sediment yields of the Khlong Phlo watershed were predicted fairly well. The result clearly implies that for this watershed, the model can be extended to study the impact of various land use change scenarios on the annual sediment yield.

3.4. Scenario analysis for impacts on water balance components

Fig. 10 depicts the change in long term annual average water balance components for single and combined biofuel crop expansion scenarios as compared to the base case.

The conversion of forest to oil palm plantation (Scenario A4) resulted in increased surface runoff by about 13% and reduced baseflow by about 7% (Fig. 10a). Similar trends can be seen for Scenario A5 (orchard, rubber and forest plantations replaced with oil palm). In contrast, for the other scenarios (A1, A2 and A3) the decrease in surface runoff and baseflow was less than 1%. Nevertheless, the change in total water yield and evapotranspiration was negligible for all scenarios analyzed.

For the scenarios of cassava expansion (Fig. 10b), there was an increase in the surface runoff and the total water yield, and a decrease in the baseflow and evapotranspiration. Among the water balance components, the change was maximum for the surface runoff for all scenarios. Comparing the scenarios, the changes are highest for B5 (orchard, rubber and forest coverage replaced with cassava). Similar trends were found for sugarcane expansion scenarios as shown in Fig. 10c.

Similar trends of results were obtained for scenarios combining two biofuel crops as can be seen in Fig. 10d. Cassava and sugarcane replacing rubber (Scenario D4) resulted in the highest change in water balance components. The least change in the water balance components was noticed for Scenario D1 (oil palm and cassava plantation replacing orchard). The combined expansion of oil palm and cassava (D1 and D2) has relatively less impacts on water

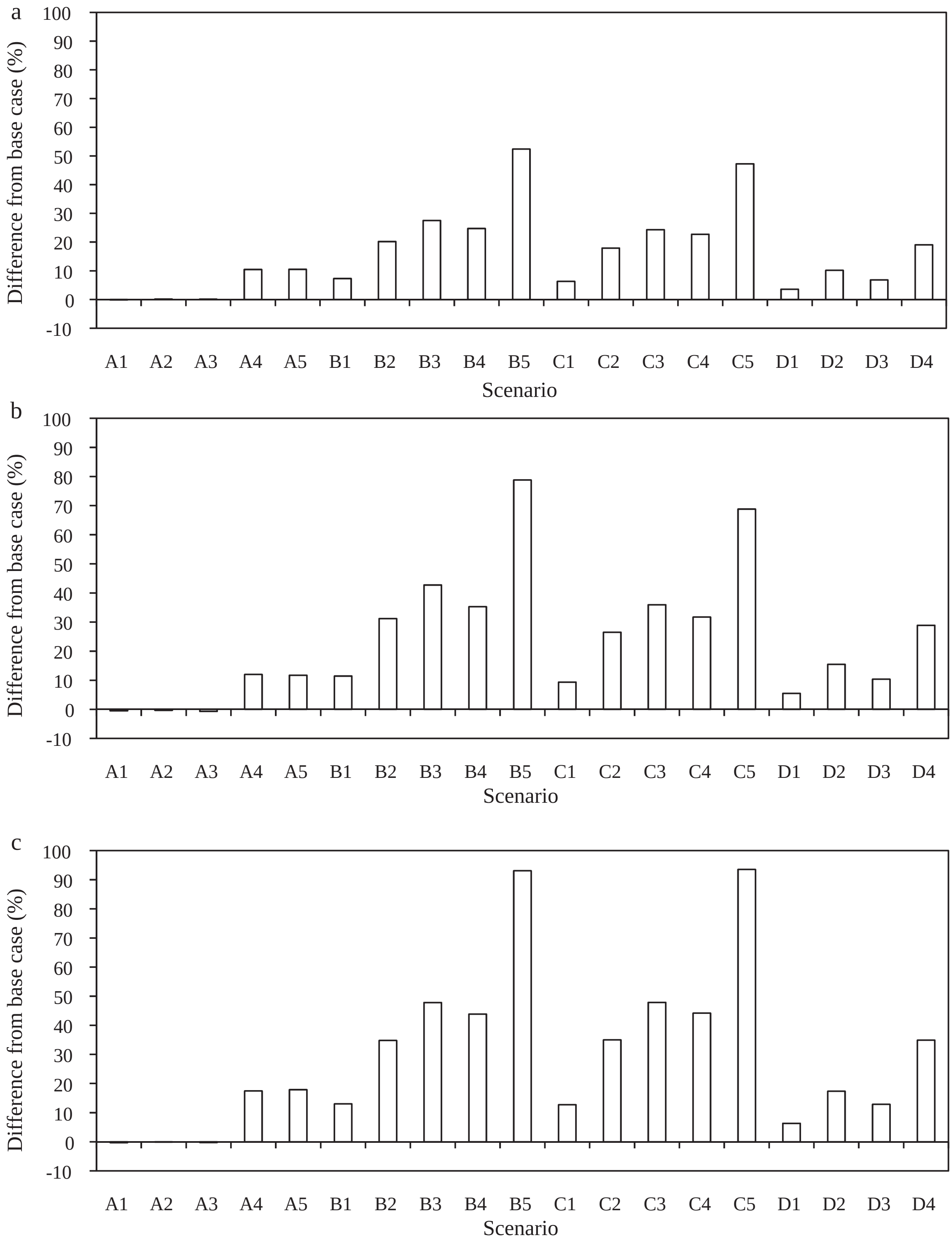

Fig. 12. Differences in annual surface runoff for different expansion scenarios compared to the base case, (a) wet year (1988); (b) normal year (1990); (c) dry year (1998).

resources compared to the expansion by only cassava replacing orchard (B1) or rubber (B2).

Analysis on monthly compared to annual basis provided interesting results for oil palm expansion as shown in Fig. 11. Although there is no change in the annual water yield for all scenarios, a significant change in monthly water yield (June, August, September and November) is found when oil palm replaces forest (Scenario A4). The cassava (B1 to B4) and sugarcane (C1 to C4) expansion will increase water yield during the wet season (May to October). During the dry season (November to April), however, no noticeable change in water yield was observed except for sugarcane for which water yield has decreased in November and December.

Fig. 12 illustrates the effects of single and combined biofuel crop expansion scenarios on the annual surface runoff in wet, normal and dry years as compared to the base case. The year with maximum rainfall was taken as wet year, average rainfall as normal year and minimum rainfall as dry year. Conversion of orchard, rubber and forest into cassava and sugarcane (Scenarios B5 and C5) results in the highest increase in surface runoff in the wet, normal and dry years. This analysis indicates that in dry years and dry areas cassava

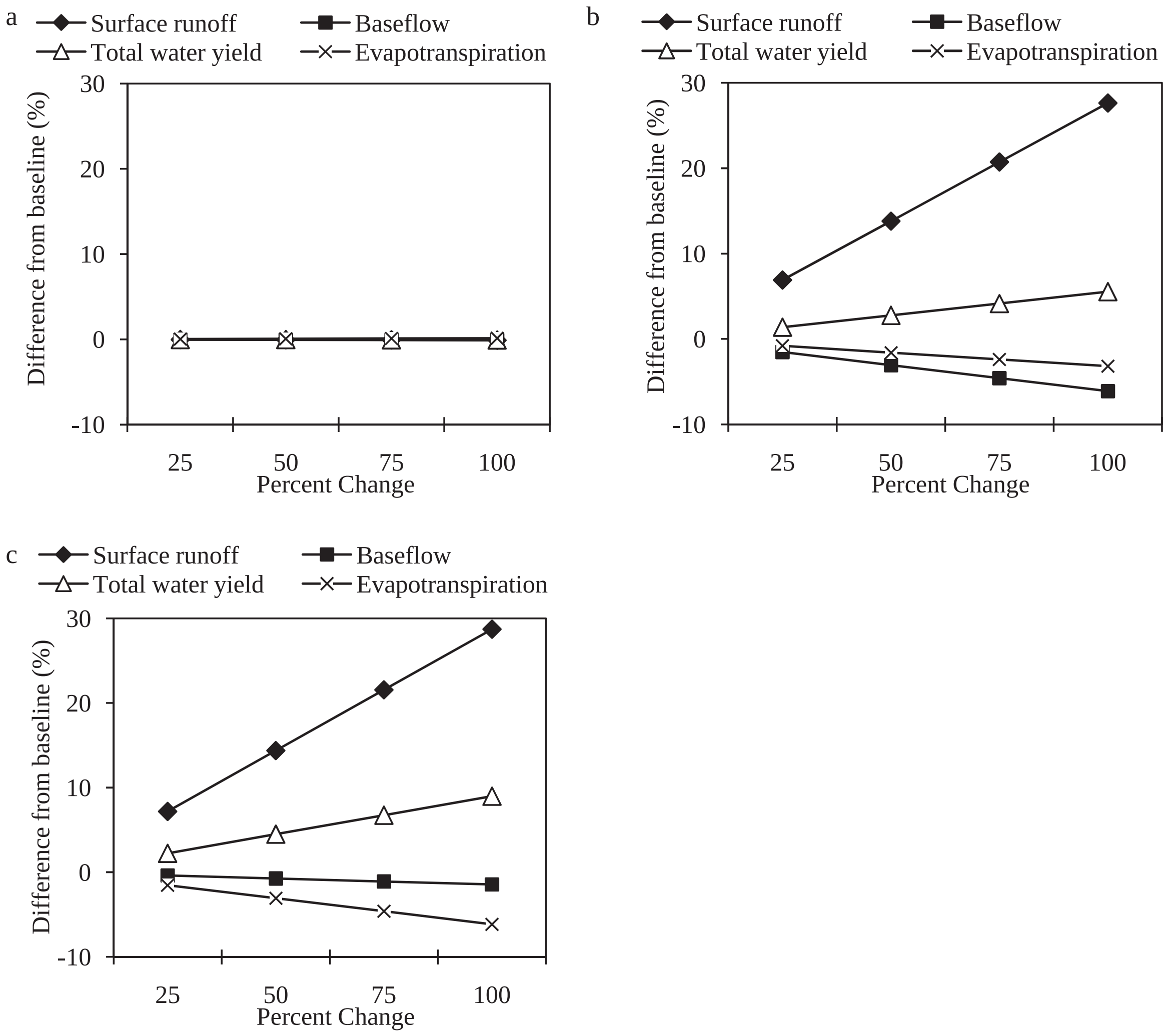

Fig. 13. Differences in annual average water balance components for different gradual expansion scenarios compared to the base case, (a) oil palm expansion; (b) cassava expansion; (c) sugarcane expansion.

and sugarcane can be grown as also reflected from the green water footprint of these two crops.

Fig. 13 shows the change in water balance components for gradual conversion of rubber cover from 0 to 100%. As expected, changes in water balance components for oil palm expansion were negligible (Fig. 13a). The surface runoff and total water yield increase and baseflow and evapotranspiration decrease with increased coverage of cassava and sugarcane (Fig. 13b and c). These results can be used to define a threshold area of biofuel crop expansion with accepted limits of change in water balance components. For example, if a 10% change in surface runoff is allowed, the threshold area for rubber conversion to cassava and sugarcane would be around 35%.

Comparing the different biofuel crops, the increase in the surface runoff was the highest for cassava expansion while the reduction in the baseflow was the highest in case of sugarcane expansion. Tree crops (like oil palm, orchard and rubber) and forest produce less surface runoff than cassava and sugarcane because of their extended roots and increased evapotranspiration (Blanco-Canqui, 2010). The change in canopy structure and surface roughness also changes the amount of evapotranspiration and the curve number which can be attributed to change in water yield (Monteith, 1965; Lahmer et al., 2001; Hu et al., 2004). Evapotranspiration can influence effective rainfall which can have significant implications on surface runoff and groundwater recharge (Stephens et al., 2001; Calder, 1993). Increased surface runoff leads to reduced infiltration and consequently a decline in the baseflow. Furthermore, the increased surface runoff would lead to a significant increase in sediment loads as well as nutrient loss.

These results clearly indicate that the conversion of orchard and/or rubber to biodiesel crops has negligible effects on water balance components on both annual and monthly basis. On the other hand, land use change for bio-ethanol production would significantly alter the annual and monthly water balance components.

Based on the results, it can be concluded that the expansion of oil palm, cassava and sugarcane should take place in areas where the current land use is orchard to minimize the impacts on the hydrological cycle. Moreover, the combined expansion of oil palm and cassava is also recommended to minimize the hydrological implication of increased biofuel production by converting rubber coverage.

3.5. Scenario analysis for impacts on water quality

Fig. 14 compares the change in annual average nitrate, total phosphorus and sediment loss for single and combined biofuel crop expansion. The nitrate loss increased for all scenarios of oil palm expansion, while the phosphorus loss increased for Scenario A1

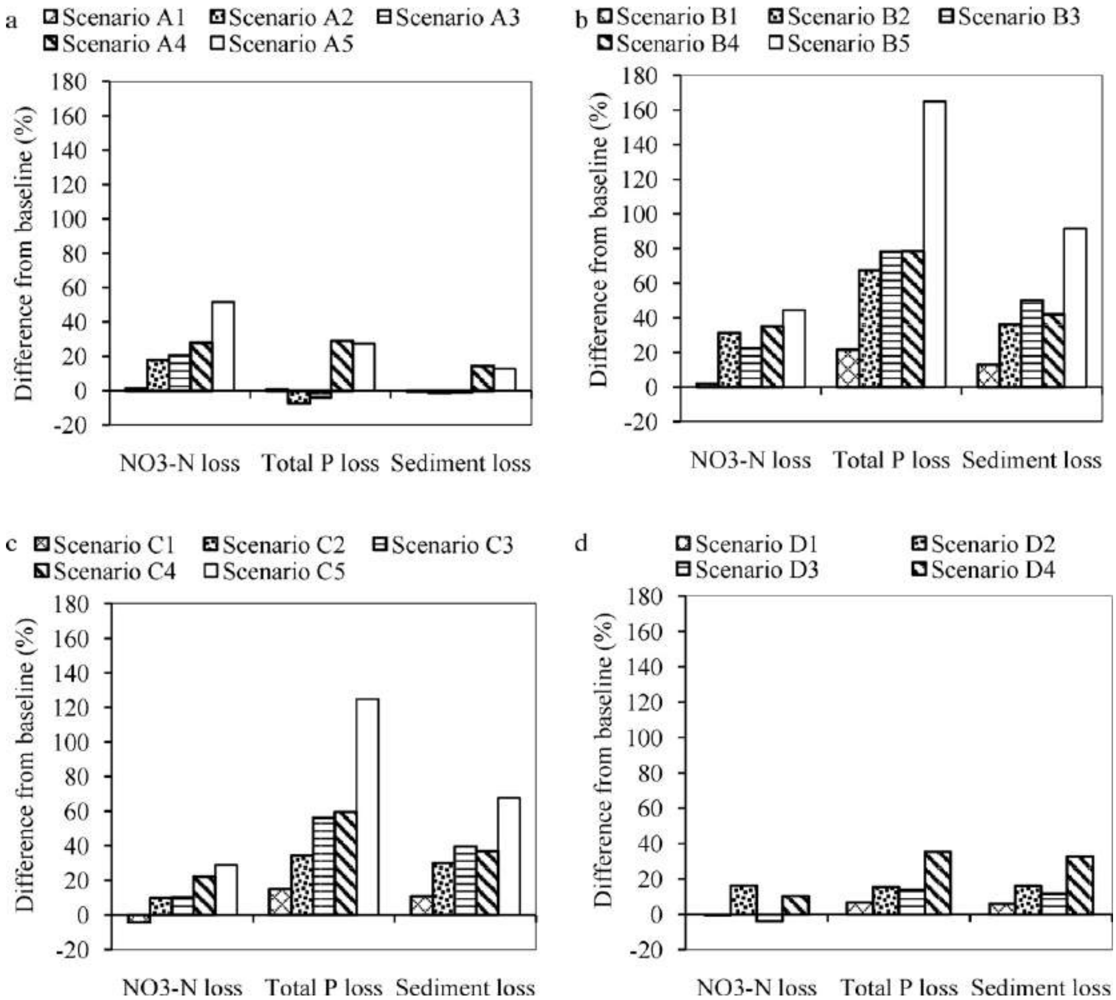

Fig. 14. Differences in non-point source pollutants for different scenarios compared to the base case, (a) oil palm expansion; (b) cassava expansion; (c) sugarcane expansion; (d) combined expansion.

(conversion of orchard to oil palm), Scenario A4 (oil palm replacing forest) and Scenario A5 (oil palm replacing orchard, rubber and forest) as shown in Fig. 14a. The sediment loss decreased for Scenarios A1, A2 and A3, except Scenarios A4 and A5 for which the sediment loss increased by nearly 15 and 13%, respectively.

The nitrate, total phosphorus and sediment loss increased for all the cassava expansion scenarios, as shown in Fig. 14b. High nitrate, phosphorus and sediment export from the watershed is evident for scenario B5 (cassava replacing orchard, rubber and forest), with an increment of about 45, 165 and 92%, respectively, with respect to the base case. Similar impact patterns can be observed for sugarcane expansion scenarios, except for Scenario C1 (orchard converted to sugarcane) where the nitrate loss was reduced by about 5%.

In Fig. 14d, it is interesting to note that the nitrate loss would be reduced by about 1% in Scenario D1 (oil palm and cassava replacing orchard) and by 4% in Scenario D3 (cassava and sugarcane replacing orchard) compared to the base case. Fig. 15 shows the difference in the annual average nitrate, total phosphorus and sediment loss compared to the base case if the gradual replacement of rubber with biofuel crops takes place. For all three crops, the nitrate loss increases with increasing coverage. The total phosphorus and sediment loss decrease with increasing oil palm coverage.

The above results suggest that expansion of cassava and sugarcane would lead to higher nitrate, phosphorus and sediment loss from the study area compared to oil palm expansion. This may be attributed to higher surface runoff and soil erosion from cassava and sugarcane areas. Higher soil losses from cassava fields compared to most other crops is due to its leaf structure which favors large drops (Moench, 1991), the wide plant spacing, the slow initial crop growth (Howler, 2001) and of its low soil coverage (approximately 50%) even at peak growth (Nguyen et al., 2008). Soil erosion tends to be high in sugarcane fields (Fiorio et al., 2000; Politano and Pissarra, 2005), in comparison to pastures and forests because extensive areas of bare soil is exposed to intense rain during the initial process of land use conversion and between crop harvesting and regrowth (Martinelli and Filoso, 2008). Soil compaction during cultivation and harvesting operations in sugarcane destroys soil physical properties such as porosity and density, which in turn decreases water infiltration and further enhances soil erosion (Fiorio et al., 2000; Prado and Centurion, 2001). The amount of nutrients lost depends mainly on the extent of erosion (Howler, 2001; Blanco-Canqui, 2010) and runoff contributes substantially to phosphorus loss (Ruppenthal et al., 1997). Further, higher erosion losses create potentially negative water quality impacts from sediments and particle bound pollutants (Thomas et al., 2009).

The replacement of forests with the biofuel crops increases pollutants in the surface water because runoff from agricultural land contains higher nutrient contents (nitrogen, phosphorous and potassium) than from forest because of higher fertilization rates used and a greater intensity of management practiced (Calder, 1998). The increased nitrate loss, because of conversion of orchard

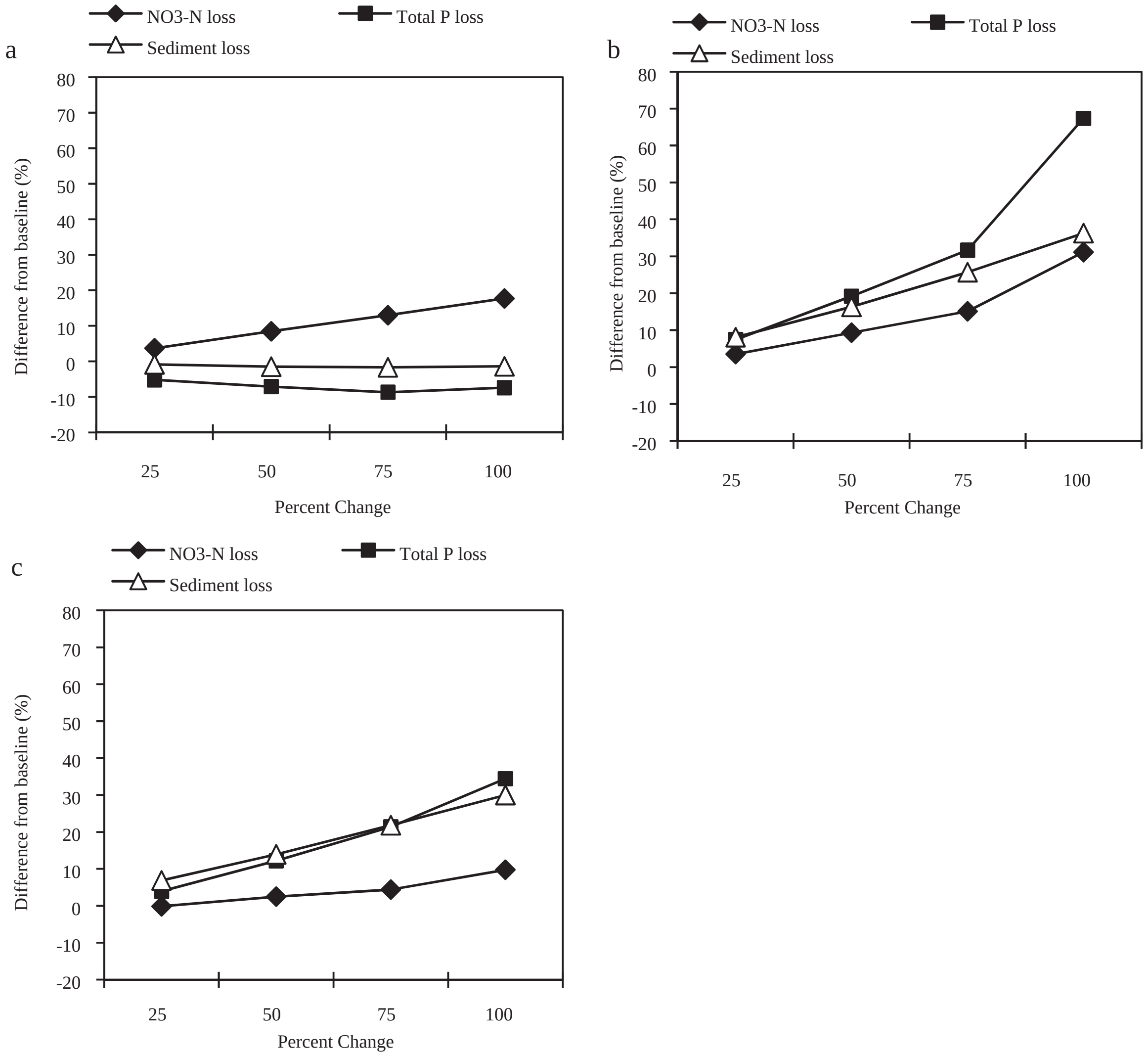

Fig. 15. Differences in non-point source pollutants for different gradual expansion scenarios compared to the base case, (a) oil palm expansion; (b) cassava expansion; (c) sugarcane expansion.

and/or rubber to oil palm, may be attributed to a higher rate of nitrogen application to oil palm. Increased concentrations of certain nutrients, particularly nitrogen and phosphorus, can promote excessive plant growth and decay in aquatic ecosystems, leading to increase in phytoplankton, decrease in dissolved oxygen, increased turbidity, loss of biodiversity, reduction in fish, increases in toxic plankton species, and other undesirable ecological effects (Smith et al., 1999; Simpson et al., 2008). The rapid growth of grain-based bio-ethanol production has already caused major water quality implications for lakes, rivers and coastal marine ecosystems in much of the USA (Thomas et al., 2008; Twomey et al., 2009) due to significant eutrophication (Simpson et al., 2008).

In summary, the expansion of oil palm for biodiesel would significantly affect the water quality except if oil palm replaces orchard which affects the water quality much less. The land use change for bio-ethanol production would undoubtedly affect the water quality due to increased sediment and nutrient loads. However, combined expansion of biofuel crops may minimize the negative implications to water quality. For example, reduced nitrate loading is expected with oil palm and cassava or cassava and sugarcane replacing orchard. An alternate approach to promote biofuel crops would be to define the threshold areas of different biofuel crops to keep the water quality within the acceptable limits. The assessment of implications of combined expansion and defining thresholds can help identify appropriate expansion plans that can be implemented to safeguard against or mitigate any potential adverse water quality consequences, which is very important for the sustainability of biofuels (Shannon et al., 2008; Engel et al., 2010). However, setting thresholds for expansion of biofuel crops from the water quality perspective is a complex issue and needs further investigations.

4. Conclusions

This study assesses the water footprint of biofuel energy from first generation biofuel crops and evaluates the impact of land use change due to the expansion of biofuel crops on the components of water balance and water quality in a small watershed, Khlong Phlo, located in the eastern part of Thailand.

Results reveal that water footprint of bio-ethanol is less than biodiesel and cassava would have less impact on water resources of the studied watershed, compared to sugarcane and oil palm. However, expansion of irrigated biofuel crops with increased use of fertilizers would lead to increased stress to freshwater resources of the watershed. The use of biofuel crops requiring less irrigation and less fertilizer would be an appropriate strategy, as also highlighted by Dominguez-Faus et al. (2009), to safeguard water resources and environment.

The SWAT model could simulate the monthly and annual water yield and the annual sediment yield very well in the studied watershed. The simulation findings indicate that the land use change from orchard and/or rubber to oil palm for biodiesel would not affect the annual and monthly water balance components. Converting forest to oil palm, however, would alter monthly water yields with no significant change in annual water yield. Expansion of oil palm area would have a significantly adverse impact on the water quality in the Khlong Phlo watershed due to increased nitrate loading into the surface water except in the case of orchard being replaced with oil palm.

On the other hand, land use change for expansion of bio-ethanol crops, cassava and sugarcane, would affect components of monthly and annual water balance with increased surface runoff and decreased baseflow and evapotranspiration. An expansion of cassava and sugarcane coverage would impact water quality due to increased sediment, nitrate and total phosphorus loadings into surface water.

Cassava is found to be most efficient crop for biofuel energy from water use view point. However, modeling results reveal that cassava expansion would affect the water quality. Hence, cassava may be promoted in water scarce areas, however, with due consideration to its impact on water quality. Simulation results clearly indicate that replacement of orchard for oil palm would have minimum impacts on both water resources and water quality which is in line with the policy of the Thai Government to promote biodiesel replacing orchards. Although conversion of rubber into oil palm would have no impact on water balance, it would affect water quality. Hence, the government of Thailand should also consider the impact on water quality before converting rubber into oil palm plantation. Since at present most of the bio-ethanol in Thailand is sugarcane based (92% in 2008), areas with abundant water resources can be appropriate for its expansion and promotion, however, water quality implication needs to be assessed. Forest in no case should be replaced for biofuel production because of its detrimental effects on water balance components and water quality.

Based on the study results, it can be recommended that orchard may be replaced for biofuel crops in Thailand from water resources and water quality perspective. Land use management plans like combined oil palm and cassava expansion and assessing threshold areas for expansion of biofuel crops should be implemented to safeguard against or mitigate any potential adverse consequences on water resources.

Since the economic dimension of biofuel production is not considered, the above recommendations are based only on the simulation results of the study. Further research on a larger scale with several options on biofuel crops, bringing into consideration the physical, socio-economic and environmental aspects, is recommended for developing suitable water and energy policies.

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