Original Research Article

Developing a Decision Support Tool for Assessing Land Use Change and BMPs in Ungauged Watersheds Based on Decision Rules Provided by SWAT Simulation

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1   **Abstract**

2   Decision making on water resources management at ungauged, especially large-scale

3   watersheds relies on hydrological modeling. Physically-based distributed hydrological

4   models require complicated setup, calibration, and validation processes, which may delay

5   their acceptance among decision makers. This study presents an approach to develop a

6   simple decision support tool (DST) for decision makers and economists to evaluate multi-

7   year impacts of land use change and BMPs on water quantity and quality for ungauged

8   watersheds. The example DST developed in the present study was based on statistical

9   equations derived from Soil and Water Assessment Tool (SWAT) simulations applied to

10   a small experimental watershed in northwest New Brunswick. The DST was

11   subsequently tested against field measurements and SWAT simulations for a larger

12   watershed. Results from DST could reproduce both field data and model simulations of

13   annual stream discharge and sediment and nutrient loadings. The relative error of mean

14   annual discharge and sediment, nitrate-nitrogen, and soluble-phosphorus loadings were -6,

15   -52, 27, and -16%, respectively, for long-term simulation. Compared with SWAT, DST

16   has fewer input requirements and can be applied to multiple watersheds without

17   additional calibration. Also, scenario analyses with DST can be directly conducted for

18   different combinations of land use and BMPs without complex model setup procedures.

19   The approach in developing DST can be applied to other regions of the world because of

20   its flexible structure.

21   **Keywords**: multiple regression; hydrological model; erosion; nitrate leaching;

22   geographic information system

1. Introduction

Pollution from nonpoint sources poses a significant threat to ecosystems and plant and animal communities (Vörösmarty et al., 2010). Nonpoint sources of sediment, nutrients, and pesticides, primarily from agricultural lands, have been identified as major contributors to water quality degradation (Zhang et al., 2004;Ongley et al., 2010). These pollutants are difficult to control because they come from many sources (Quan and Yan, 2001). Practices such as strip cropping, terracing, crop rotation, and nutrient management can be developed to prevent soil erosion and reduce the movement of nutrients and pesticides from agricultural lands to aquatic ecosystems (D'Arcy and Frost, 2001). These pollution-prevention methods, known as best management practices (BMPs), are intended to minimize the negative environmental impact of agricultural activities, while maintaining land productivity. Reliable information on the impacts of land use change and BMPs on water quantity and quality is critical to watershed management (Panagopoulos et al., 2011).

Many studies have been conducted to evaluate the impact of land use change and BMPs on water quality based on field experiments (Novara et al., 2011;Pimentel and Krummel, 1987;Sadeghi et al., 2012;Turkelboom et al., 1997;Urbonas, 1994). Monitoring systems have been established to assess the impact of land use change and BMPs on water resources in order to capture the spatial and temporal variation in soil, climate, and topographic conditions in watersheds (Veldkamp and Lambin, 2001). Statistical models developed from field data from small watersheds are usually assumed to apply to large watersheds (Blöschl and Sivapalan, 1995;Bloschl and Grayson, 2001). Although it is not difficult to quantify soil erosion and chemical loadings in experimental plots, it is time-

consuming and expensive (Mostaghimi et al., 1997). Clearly, it is not practical to conduct field experiments for every possible combination of land use and BMPs, under different biophysical conditions. As a result, it is unlikely sufficient field data could be obtained to develop management plans and conduct cost-benefit analyses. In addition, statistical models could be potentially derived from experiments; however, it is difficult to establish cause-and-effect relationships between BMPs and water quality variables under varied biophysical conditions or to quantify the impact of combined land use and BMPs on water quality at the watershed scale (Renschler and Lee, 2005).

Process-based models of hydrology can be used to extrapolate field data to fill data gaps (Borah and Bera, 2004;Borah and Bera, 2003;Singh, 1995;Singh and Woolhiser, 2002;Singh and Frevert, 2005). These process-based models provide quantitative information that is usually difficult to obtain from field experiments (Borah et al., 2002). For example, ANSWERS (Beasley et al., 1980), CREAMS (Knisel, 1980), GLEAMS (Leonard et al., 1987), AGNPS (Young et al., 1989), EPIC (Sharpley and Williams, 1990), and SWAT (Arnold et al., 1998) have been used to understand surface runoff, soil erosion, nutrient leaching, and pollutant-transport processes. However, these process-based models require extensive input data and complex calibration procedures (Liu et al., 2015); watersheds with sufficient data to calibrate and validate these models are normally small, resulting in lack of representation at large spatial scales. Furthermore, once a model is calibrated, parameters become watershed-specific, which cannot be easily extended to other watersheds. In addition, these models require specialized expertise, which prevents non-expert decision makers and economists to use them (Viavattene et al., 2008).

70   A decision support tool could be developed by combining "decision rules" with

71   geographic information systems (GIS) for water quality assessment in large ungauged

72   watersheds. The "decision rules" could be based on regression equations derived from

73   field experiments (Renschler and Harbor, 2002), or they could be defined simply as

74   constants based on expert knowledge. Alternatively, simulations from a well-calibrated

75   hydrological model could be used to develop statistical equation-based "decision rules".

76   Apart from defining "decision rules" at each grid cell, to assess water quantity and

77   quality in streams or at subbasin/watershed outlets, the decision support tool should

78   consider discharge, sediment, and nutrient routing within the watershed. For example, a

79   commonly used routing mothed for sediments is the sediment-delivery ratio (SDR)

80   method, which is widely employed in many GIS-based erosion models (May and Place,

81   2010;Wilson et al., 2001;Zhao et al., 2010). For discharge, a simple summation routing at

82   the outlet produces acceptable accuracy for small- and medium-sized watersheds,

83   considering that there is negligible water losses from surface runoff and stream flow. For

84   large watersheds, water losses are generally greater. These water losses can be estimated

85   using simple linear equations. The annual export of nutrients from watersheds (via the

86   nutrient-delivery ratio) has been studied empirically in many studies as nutrient loading

87   per land area (Endreny and Wood, 2003;Beaulac and Reckhow, 1982;Reckhow and

88   Simpson, 1980).

89   A decision support tool developed based on "decision rules" is generally flexible and

90   easy for decision makers and economists to use (Endreny and Wood, 2003). However,

91   their practicality in normal circumstances, particularly with respect to their level of

92   accuracy, needs to be evaluated. In addition, to provide sufficient "decision rules" with

93   reasonable accuracy, fully validated hydrological models are required to be able to fill

94   data gaps in field experiments. The present study used the Soil and Water Assessment

95   Tool (SWAT) to provide modelled data in the development of the decision support tool.

96   The main objective of the present study is to develop a simple decision support tool with

97   the intent to evaluate the impact of land use change and BMPs on water resources in a

98   large ungauged watershed in New Brunswick, Canada. This paper presents the

99   development and testing of a decision support tool using data from two watersheds in the

100   potato-belt of New Brunswick; one small experimental watershed, with extensive

101   monitoring and field survey data, and a larger watershed containing the smaller

102   watershed. Specifically, this involves: (1) setting up, calibrating, and validating SWAT

103   for a small experimental watershed; (2) developing statistical equations relating water

104   quality and quantity variables with weather, soil, land use information based on SWAT

105   simulations for different combinations of land use and BMPs; (3) integrating the

106   statistical equations into a decision support tool with the aid of ArcGIS; and (4) testing

107   the decision support tool against field measurements and model simulations of stream

108   discharge, sediment, and nutrient loadings for a large watershed.

109   **2. Materials and Methods**

110   **2.1 Study Sites and Data Collection**

111   The large watershed of this study is the Little River Watershed (LRW), located in the

112   Upper Saint John River Valley of northwestern New Brunswick, Canada (Fig. 1). It

113   covers an area approximately 380 km^2 with a mixture of agricultural (16.2%), forest

114   (77%), and residential (6.8%) land uses (Xing et al., 2013). Elevation in the watershed

115   ranges from 127 to 432 m above mean sea level (Fig. 1). The soil in the study sites is

116   classified as mineral, derived from various parent materials. The major associations are

117   Caribou, Carleton, Glassville, Grandfalls, Holmesville, McGee, Muniac, Siegas, Thibault,

118   Undine, Victoria, Waasis, and one organic soil (Fig. 2). The study site belongs to the

119   Upper Saint John River Valley Ecoregion in the Atlantic Maritime Ecozone (Marshall et

120   al., 1999). The climate of the region is considered to be moderately cool boreal with

121   approximately 120 frost-free days, annually (Yang et al., 2009). Daily maximum and

122   minimum temperate are 24 (in July) and -18.1˚C (in January) based on Canadian Climate

123   Normal station data at St. Leonard (http://climate.weather.gc.ca/climate_normals). The

124   average temperature is 3.7˚C and annual precipitation is 1037.4 mm (Zhao et al., 2008).

125   About one-third of the precipitation is in the form of snow. Snowmelt leads to major

126   surface runoff and groundwater recharge events from March to May (Chow and Rees,

127   2006). The land use and soil maps in the setup of SWAT for LRW were derived from

128   publicly available data [Energy and Resource Development (ERD), New Brunswick; Fig.

129   2].

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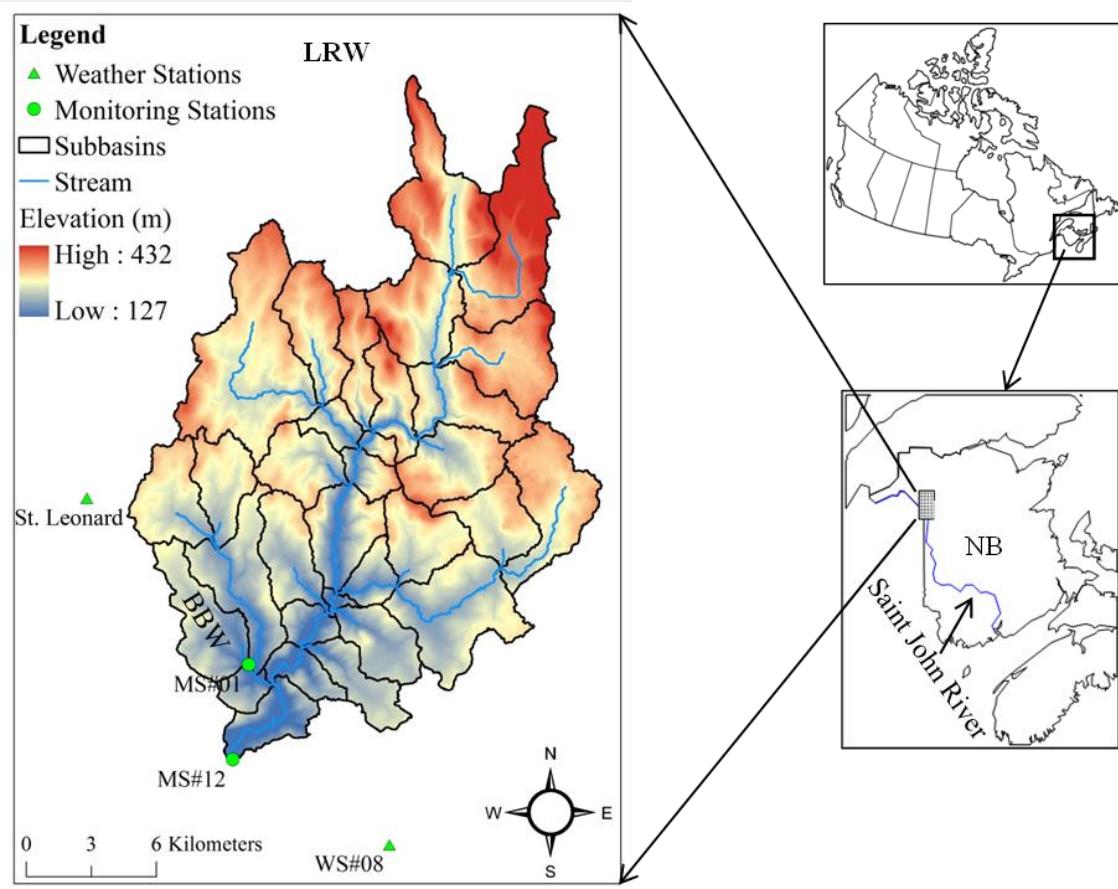
Fig. 1 Location of the Little River Watershed (LRW) and Black Brook Watershed (BBW)

in New Brunswick (NB), Canada and water-monitoring stations #01 and #12 as well as

weather stations #08 and St. Leonard. Elevations and subbasins are also shown for LRW.

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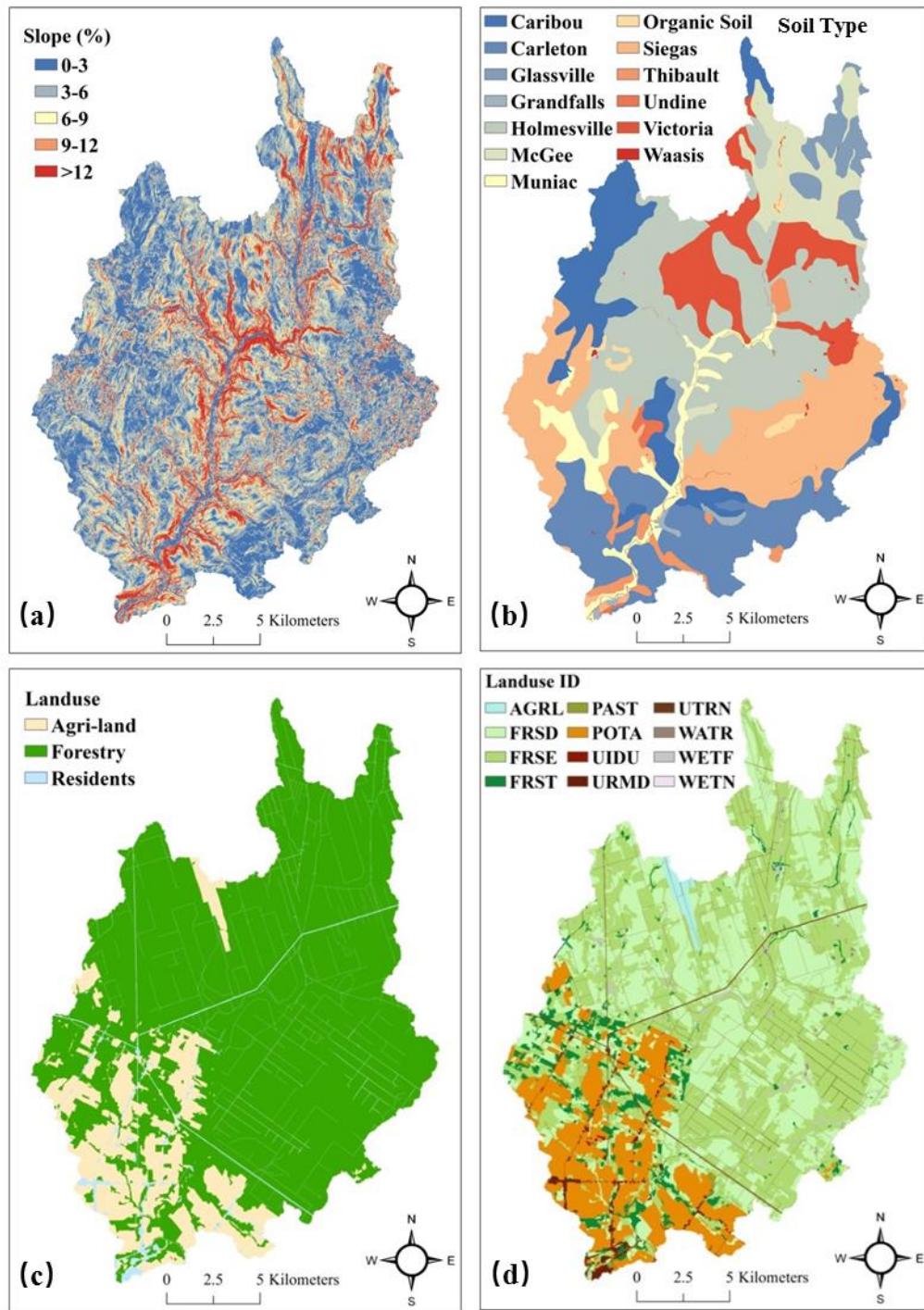
Fig. 2 Slope classes created using a 10-m resolution LiDAR (Light Detection and

Ranging)-based DEM (Digital Elevation Model), soil and land use maps, and land use

IDs in SWAT (see Table 2 for land use ID meaning).

The small experimental watershed of the study is the Black Brook Watershed (BBW),

a subbasin of LRW (Fig. 1). The BBW has been studied extensively for more than 20

years to evaluate the impact of agriculture on soil erosion and water quality (Li et al.,

2014;Chow and Rees, 2006). The watershed covers an area of 14.5 km^2, with 65% being

agriculture land, 21% forest land, and 14% residential areas and wetlands. Slopes vary

from 1-6% in the upper basin to 4-9% in the central area. In the lower portion of the

watershed, slopes are more strongly rolling at 5-16%. Soil surveys (1:10,000 scale)

identified six mineral soils, namely Grandfalls, Holmesville, Interval, Muniac, Siegas,

and Undine, and one organic soil, St. Quentin (Mellerowicz, 1993).

A water-monitoring station was established at the outlet of BBW in 1992 (MS#01; Fig.

1) and another (MS#12) at the outlet of LRW in 2001. At these stations, V-notch weirs

were installed, and the stage height of the water was recorded using a Campbell-

Scientific CR10X data logger. Stage height values were converted to total flow rates with

a calibration curve function (Chow et al., 2011). Water samples were collected with an

ISCO automatic sampler. Sampling frequency was set at one sample every 72 hours when

runoff was absent. During runoff events, sampling frequency was increased to one

sample every 5-cm change in stage height. Samples were analyzed for concentration of

suspended solids, nitrate-nitrogen (NO_3-N), and soluble-phosphorus (Sol-P). Detailed

description of data collection procedures and sample analyses can be found in Chow et al.

(2011). Weather data including daily precipitation, air temperature, relative humidity, and

wind speed were acquired from the St. Leonard Environment Canada weather station

(http://climate.weather.gc.ca), located approximately 5 km northwest of BBW (Fig. 1).

The daily average relative humidity and wind speed were calculated based on averaging

163   hourly values. Since this weather station did not monitor daily solar radiation, the study

164   used solar radiation collected from a weather station located approximately 10 km

165   southeast of BBW (WS#08; Fig. 1).

166   **2.2 SWAT Setup, Calibration, and Validation for BBW and LRW**

167   A modified version of SWAT has been developed for cold regions (Qi et al.,

168   2017a;Qi et al., 2016a, b;Qi et al., 2017b), and it was used for the BBW and LRW in this

169   study. Detailed model setup, calibration, and validation for BBW can be found in Qi et al.

170   (2017b). Specific model inputs for both watersheds are provided in Table 1. The same

171   weather data were used for both watersheds (Table 1). The Digital Elevation Model

172   (DEM) for LRW and BBW were both based on high resolution LiDAR (Light Detection

173   and Ranging) data, the first was created at 10-m and the second, at 1-m resolution. The

174   LRW was delineated into 32 subbasins from which their topographic characteristics were

175   defined (Fig. 1). The soil types and slopes, which were classified into five separate

176   classes, are illustrated in Fig. 2 for LRW. After combining the soil, slope, and land use

177   maps through the ArcSWAT-interface function, 362 HRUs were subsequently created for

178   LRW (based on thresholds: 10, 15, and 20% for land use, soil, and slope).

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186   **Table 1** Datasets in SWAT setup, calibration, and validation for BBW and LRW.

Dataset	BBW	LRW
LiDAR DEM resolution	1-m	10-m
Soil map	Survey (1993)	ERD
Land use maps	Survey (1992-2011)	ERD (one map)
Precipitation, temperature, relative humidity & wind speed	St. Leonard (1992-2011)	St. Leonard (2001-2010)
Solar radiation	WS#08 (1992-2011)	WS#08 (2001-2010)
Contour tillage operation (spring and fall)	Survey (1992-2011)	Only for potato and barley (2001-2010)
Fertilizer application	Survey (1992-2011)	Estimated from BBW (2001)
Crop rotation	Survey (1992-2011)	Potato-barley (2001-2010)
Terraces and grassed waterways	Survey (1992-2011)	Negligible
Discharge, sediment, NO_3-N and Sol-P	MS#01 (1992-2011)	MS#12 (2001-2010)

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188   Since only one land use map was available for LRW (Table 1), assumptions were

189   made based on information available on land use and management records for BBW to

190   adjust the SWAT-management files for LRW as follows:

191   (1) Potato-barley rotations were assigned to the land use ID POTA (Table 2); for other

192   land use IDs, a single crop was considered;

193   (2) Fertilizers were applied only to potato and barley fields, and fertilizer amounts and

194   N:P (nitrogen-to-phosphorus) ratios were averaged for potato and barley fields over the

195   entire watershed, based on 2001 survey data from BBW;

196   (3) Contour tillage was applied only to potato and barley fields;

197   (4) Flow diversion terraces (FDT) and grassed waterways in LRW were assumed not

198   used. It is worth noting that these four assumptions serve as a baseline scenario for the

199   assessment of FDT in LRW.

200   To evaluate the global performance of the decision support tool for LRW, related

201   land use and management files were prepared and accessed by SWAT. For purpose of

202   comparison, simulations with SWAT were produced in an initial application by setting

203   the adjustable parameters of the model to their default values, and in a second application

204   by setting the parameters according to values produced with a watershed-specific model

205   calibration to BBW. This approach with model parameterization is widely accepted when

206   applying SWAT to large ungauged watersheds (Panagopoulos et al., 2011).

207   **2.3 Decision Rules**

208   The decision support tool was designed to use the "decision rules" to estimate annual

209   discharge and sediment and nutrient loadings from individual grid cells:

210    $A = \sum_{i=1}^{n} DR_i \cdot A_i,$ (1)

211   where A is the annual discharge or sediment and nutrient loadings at the outlet of the

212   watershed, DR_i and A_i are the delivery ratios and annual discharge or loadings,

213   respectively, for grid cell i. For the present study, statistical equations derived from

214   simulations of the calibrated version of SWAT for BBW were defined as the "decision

215   rules" in the decision support tool.

216   **2.3.1 Land Use Groups and BMP Scenarios**

217   In statistical equation development, land use in BBW (24, in total) was first classified

218   into five land use classes according to their influences on hydrological processes (Table

219   2). Note that WATR was not used due to its small overall coverage (Fig. 2). As for

220   watershed management, we considered three main BMPs, i.e.,

221   (1) FDT + contour tillage;

222   (2) Contour tillage; and

223   (3) No-BMP (without FDT and contour tillage).

224

225   **Table 2** Land use and land use groups (LUGP) for BBW and LRW.

LUGP	Land use ID in SWAT	Land use type
AGRL	AGRL	Agricultural Land-Generic
(General crops)	CANA	Canola
	CRON	Corn
	FPEA	Field peas
	POTA	Potato
GRAN	BARL	Barley
(Grains)	OATS	Oats
	PMIL	Millet
	RYE	Rye
	SWHT	Spring wheat
	WWHT	Winter wheat
GRAS	BERM	Bermuda grass
(Grasses)	CLVR	Clover
	HAY	Hay
	PAST	Past
	RYEG	Ryegrass
	TIMO	Timothy
FORT	FRSD	Forest-Deciduous
(Forestry)	FRSE	Forest-Evergreen
	FRST	Forest-Mixed
	RNGB	Range-Bush
	WETF	Wetlands-Forested
	WETN*	Wetlands-No-Forest
NOCR	URMD	Residential
(Non-vegetated	UTRN	Transportation
lands)	UIDU*	Industrial

Note: "*" indicates unique land use types to LRW not present in BBW and, therefore,
unaccounted for in the development of the decision support tool.

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227   The calibrated version of the enhanced SWAT for BBW was used to generate annual

228   outputs based on HRUs from 1992 to 2011. The model was run three times to generate

229   the BMP-specific data for statistical equation development.

230

231   **2.3.2   Explanatory Variables Selection**

232   Explanatory candidate variables must be physically-meaningful in hydrological and

233   biochemical processes. It is worth noting that both continuous and categorical variables

234   were included in the regression equation. The land use group (LUGP) was the only

235   categorical variable, and the remaining were all continuous variables. To detect

236   significant predictors, the analysis of covariance (ANCOVA) was used. It requires at

237   least one continuous and one categorical explanatory variable and is used to identify the

238   major and interaction of predictor variables. By including continuous variables, the

239   method can reduce the variance of error to increase the statistical power and precision in

240   estimating categorical variables (Keselman et al., 1998;Li et al., 2014). Inclusion of

241   interaction terms in these regression models dramatically increased model performance.

242   In the present study, we only considered interactions between two explanatory variables

243   at a time. Student t-tests were conducted to examine the statistical significance of each

244   level of LUGP and their interaction with the various continuous variables. When one

245   level of LUGP (e.g., GRAN; Table 2) did not significantly correlate with water quality or

246   quantity, or there were nominal interactions between a given level and other explanatory

247   variables, this particular level of LUGP would be combined with other levels of LUGP

248   until all new levels of LUGP were statistically significant.

249   Multiple linear regression analyses were used to relate annual total discharge (mm) and

250   sediment (t ha^{-1}), NO_3-N (kg ha^{-1}), and Sol-P (kg ha^{-1}) loadings to the explanatory

251   variables. These work was conducted in R (Ihaka and Gentleman, 1996). Only six

252   continuous explanatory variables were determined for the specification of the statistical

253   models. Annual precipitation (PCP), annual mean air temperature (TMP), and mean

254   saturated hydraulic conductivity of soil (SOL_K) were common to the dependent

255   variables (i.e., total discharge and sediment, NO_3-N, and Sol-P loadings). The LS-factor

256   (USLE_LS) and annual N and P application rates (N_APP and P_APP) were unique to

257   the equations addressing sediment, NO_3-N, and Sol-P loading.

258   2.3.3   Delivery Ratio Definition

259   The LS-factor of the universal soil loss equation (USLE) was determined by slope

260   gradient (*slp*) and slope length (*L*) of individual HRUs:

261     $$\text{USLE\_LS} = \left\{\frac{L}{22.1}\right\}^m \cdot (65.41 \cdot sin^2(a) + 4.56 \cdot \sin(a) + 0.065) \qquad (2)$$

262   where *m* is the equation exponent and *a* is the angle of the slope (in degrees). The

263   exponent *m* is calculated by,

264     $$m = 0.6 \cdot (1 - \exp[-35.835 \cdot slp]) \qquad (3)$$

265   where *slp* is in units of m m^{-1}. For the decision support tool, slope length *L* equals to the

266   length of the grid side and slope gradient was determined by the *Slope* tool in ArcGIS.

267   The sediment-delivery ratio was not considered in the decision support tool application to

268   BBW. We assumed that annual sediment loadings from grid cells in decision support tool

269   were all exported to the outlet of BBW. However, when the decision support tool was

270   applied to LRW, the sediment-delivery ratio was used to correct estimates of sediment

271   loading at the watershed outlet. The sediment loadings at the outlet of LRW (*sed*) were

272   determined by

273     $$sed = SDR \cdot sed\tilde{} \qquad (4)$$

274   where *sed~* is the sediment loading calculated with the sediment loading equation (one for

275   each BMP and land use group), and *SDR* is determined by (Vanoni, 1975)

276     $$SDR = 0.37 \cdot D^{-0.125} \qquad (5)$$

277   where D (km^{-2}) is the drainage area. For annual discharge and nutrient loadings, we

278   assumed their delivery ratios equal to 1.0 for all grid cells in LRW.

279   **2.4 Decision Support Tool Assessment**

280   Inputs to the decision support tool included the six continuous explanatory variables

281   and LUGP as well as information on management practices, e.g., contour tillage and FDT

282   implementation. Simulations from each grid cells were summarized at the outlet of the

283   study watersheds. We first tested the impact of cell size on simulations of water quantity

284   and quality at the outlet of BBW. The cell size range was determined by considering

285   different farmland sizes in the watershed. We assumed that farmland-based grid cells can

286   sufficiently represent basic hydrological processes, land use change, and management

287   practice implementations for hydrological modeling. Simulated annual water flow and

288   sediment and nutrient loadings with the decision support tool were compared with those

289   produced with the calibrated version of the enhanced SWAT. Subsequently, the decision

290   support tool was applied to LRW, and the simulations were compared with the results of

291   the uncalibrated and calibrated versions of SWAT. The purpose of this was to test if the

292   decision support tool (i.e., land use and BMP assessment tool; LBAT) performed better,

293   or at least as well, as both the uncalibrated and calibrated version of SWAT.

294   Model performance in terms of water quantity and quality at the outlet of the study

295   watersheds was assessed based on the coefficient of determination (R^2) and relative error

296   (Re), i.e.,

297    $$R^2 = \left(\frac{\sum_{i=1}^n (O_i - O_{avg}) \cdot (P_i - P_{avg})}{\left[\sum_{i=1}^n (O_i - O_{avg})^2 \cdot \sum_{i=1}^n (P_i - P_{avg})^2 \right]^{0.5}} \right)^2 \tag{6}$$

298    $$Re = \frac{(P_{avg} - O_{avg})}{O_{avg}} \cdot 100\% \tag{7}$$

299   where O_i, P_i, O_{avg}, and P_{avg} are the observed and predicted and averages of the observed

300   and predicted values, respectively.

301

302   **2.5 FDT Assessment in LRW**

303   A series of FDT-implementation scenarios were set up for LBAT based on six slope

304   classes to assess the impact of FDT on water quantity and quality on agricultural lands in

305   LRW (Fig. 3; Table 3). From scenarios one (S1) to six (S6), total area protected by FDT

306   gradually increased until all agricultural lands were protected (Table 3). Mean annual

307   simulations of total discharge and sediment, NO_3-N, and Sol-P loadings from LRW from

308   2001 to 2010 were compared with those of the baseline scenario (FDT = 0%) for each

309   scenario using two performance indicators, i.e., mean difference (MD) and % relative

310   difference (PRD), given as:

311   (1) MD = output with FDT − output without FDT, and

312   (2) PRD (%) = MD/output without FDT × 100.

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314   **Table 3** Slope classes and corresponding areas in the agricultural land of LRW.

Scenario	Slope	Area protected by FDT (ha)	Agricultural lands (%)
S1	≥5%	624	10
S2	≥4%	1328	22
S3	≥3%	2224	37
S4	≥2%	3680	61
S5	≥1%	5360	89
S6	≥0	6048	100

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317 **3. Results and Discussion**

318 **3.1 Statistical Equations (Decision Rules)**

319 **3.1.1 Model Structure and Coefficients**

320   Linear regression equations and their explanatory variables for annual discharge and

321 sediment, NO_3-N, and Sol-P loadings under different combinations of land use groups

322 and BMP scenarios are provided in Tables 4 and 5. In total, three discharge models (Dis1,

323 Dis2, and Dis3) and five sediment (Sed1_1, Sed1_2, Sed1_3, Sed2, and Sed3), NO_3-N

324 (N1_1, N1_2, N1_3, N2, and N3), and Sol-P (P1_1, P1_2, P1_3, P2, and P3) loading

325 models were developed. Data transformations (via logarithm and power transformations)

326 were applied to sediment, NO_3-N, and Sol-P loadings to meet the assumption of

327 normality in multiple regression analysis (Table 4). The contour tillage and FDT were

328 applied only to agricultural lands, including land use groups AGRL, GRAN, and GRAS

329 (Table 4). For the no-BMP scenario, three separate sediment, NO_3-N, and Sol-P loading

330 models were developed for agricultural lands (AGRL, GRAN, and GRAS), non-

331 vegetated lands (NOCR), and forest lands (FORT), and one discharge model (Dis1) for

332 all land use groups (Table 4). It is worth noting that the sediment loading model, Sed3,

333 was a modified version of Sed1_1 (multiplied by TERR_P) for the FDT + contour tillage

334 scenario (Table 4), and the values of TERR_P (Qi et al., 2017b) used for Sed3 were the

335 same as the calibrated values in SWAT for BBW (Qi et al., 2017b). Also, NO_3-N and

336 Sol-P loadings (N1_2 and P1_2) for non-vegetated lands (NOCR) were determined as

337 constants, which were equal to the calculated means of NO_3-N and Sol-P loadings

338 determined by SWAT (i.e., 24 and 0.61 kg ha^{-1}, respectively; Table 4).

339   As for LUGP (including AGRL, GRAN, GRAS, FORT, and NOCR; Table 2), three

340   new land use groups (i.e., LUGP1, LUGP2, and LUGP3) were formulated by combining

341   agricultural lands AGRL, GRAN, and GRAS during model development (Tables 4 and 5).

342   For example, LUGP2 was derived by combining AGRL, GRAN, and GRAS on total

343   discharge (i.e., Dis1 model). Individual model structures are shown in Table 4, whereas

344   the explanatory variables for these models appear in **Appendix A**. The coefficients

345   estimated for the explanatory variables and their interactions, and their t-test results are

346   also shown in **Appendix A**. Most of the *p*-values for these explanatory variables were <

347   0.001, except for several that were between 0.001 and 0.08, which were also taken as

348   acceptable.

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Table 4 Statistical models based on land use groups (LUGP) and BMPs.

BMPs	LUGP*	Model	Structure
No-BMP	CRGP2,NOCR,FORT	Dis1	Discharge $= f$ (PCP, TMP, SOL_K, LUGP2)
Contour tillage	AGRL,GRAN,GRAS	Dis2	$= f$ (PCP, TMP, SOL_K)
FDT+Contour tillage	AGRL,GRAN,GRAS	Dis3	$= f$ (PCP, TMP, SOL_K)
No-BMP	CRGP1,GRAS	Sed1_1	Sediment$^{(1/10)} = f$ (USLE_LS, PCP, TMP, SOL_K, LUGP1)
	NOCR	Sed1_2	$= f$ (USLE_LS, PCP)
	FORT	Sed1_3	$= f$ (USLE_LS, PCP, SOL_K)
Contour tillage	CRGP1,GRAS	Sed2	Sediment$^{(1/10)} = f$ (USLE,_LS, PCP, TMP, SOL_K, LUGP1)
FDT+Contour tillage	AGRL,GRAN,GRAS	Sed3	Sediment $=$ Sed1_1 $\times$ TERR_P
No-BMP	AGRL,GRAN,GRAS	N1_1	Log(NO$_3$-N) $= f$ (N_APP, PCP, TMP, SOL_K, LUGP)
	NOCR	N1_2**	NO$_3$-N$= 24$ kg ha^{-1}
	FORT	N1_3	Log(NO$_3$-N) $= f$ (PCP, TMP, SOL_K)
Contour tillage	AGRL,GRAN,GRAS	N2	Log(NO$_3$-N) $= f$ (N_APP, PCP, TMP, SOL_K, LUGP)
FDT+Contour tillage	CRGP3,GRAN	N3	$= f$ (N_APP, PCP, TMP, SOL_K, LUGP3)
No-BMP	CRGP1,GRAS	P1_1	Log(Sol-P) $= f$ (P_APP, PCP, TMP, SOL_K, LUGP1)
	NOCR	P1_2**	Sol-P $= 0.61$ kg ha^{-1}
	FORT	P1_3	Log(Sol-P) $= f$ (PCP, TMP, SOL_K)
Contour tillage	CRGP1,GRAS	P2	Log(Sol-P) $= f$ (P_APP, PCP, TMP, SOL_K, LUGP1)
FDT+Contour tillage	AGRL,GRAN,GRAS	P3	$= f$ (P_APP, PCP, TMP, SOL_K, LUGP)

*AGRL and GRAN are combined into one group, namely CRGP1 in LUGP1; AGRL, GRAN and GRAS are combined into one group, namely

CRGP2 in LUGP2; AGRL and GRAS are combined into one group, namely CRGP3 in LUGP3; ** variable is set constant.

355 **Table 5** Explanatory variables determined for statistical analysis.

Variable	Unit	Meaning
LUGP	—	Land use groups including AGRL, GRAN, GRAS, FORT, and NOCR
LUGP1	—	AGRL and GRAN are combined into a new group, CRGP1
LUGP2	—	AGRL, GRAN, and GRAS are combined into a new group, CRGP2
LUGP3	—	AGRL and GRAS are combined into a new group, CRGP3
N_APP	kg ha^{-1}	Annual N application rate
P_APP	kg ha^{-1}	Annual P application rate
PCP	mm	Annual precipitation
SOL_K	mm h^{-1}	Mean saturated hydraulic conductivity of soil
TERR_P	—	P-factor for FDT
TMP	℃	Annual mean air temperature
USLE_LS	—	LS-factor of USLE

356

3.1.2 Statistical Equation Assessment

358   Simulations based on the statistical equations and the calculated outputs from

359 individual HRUs for the different BMPs are compared in Table 6. In general, discharge

360 models were able to reproduce SWAT simulations for the three BMPs; R^2 ranging from

361 0.86 to 0.9. Mean discharge simulated with the statistical equations was equal to that of

362 SWAT (Table 6). Mean discharge (636 mm) for the no-BMP-case (BMP 3) was greater

363 than that for BMPs using contour tillage and FDTs (619 and 628 mm for BMP 1 and 2,

364 respectively), suggesting that contour tillage and FDTs can cause evapotranspiration to

365 increase.

366   Models Sed1_2 and Sed1_3 were able to reproduce simulations with SWAT (yielding

367 $R^2 = 0.71$ and 0.57, respectively), and simulated mean sediment loadings were close to

368 that of SWAT (Table 6). Models Sed1_1 and Sed2 tended to underestimate results from

369 SWAT (Table 6), with an overall lower mean sediment loading of 10.78 vs. 12.84 and

370 8.31 vs. 9.4 t ha^{-1}, respectively. Mean sediment loading with Sed3 (0.89 t ha^{-1}) was

371 slightly greater than that of SWAT (0.84 t ha^{-1}), because Sed3 only took into account

372   TERR_P, whereas SWAT took into account TERR_CN and the impact of grassed

373   waterways. Results from the statistical equations showed that the mean sediment loading

374   for BMP 2 (8.31 t ha^{-1}) was significantly different than that for BMPs 1 and 3, with mean

375   loading of 0.89 and 10.78 t ha^{-1} (Table 6). The smallest mean sediment loading (0.09 t ha^{-1}

376   ) was found to occur with the FORT land use grouping (Table 6).

377   The four NO_3-N and Sol-P loading equations explained ~50% of the variation in the

378   SWAT simulations for the same variables, with R^2 ranging from 0.33 to 0.59 (Table 6).

379   Mean NO_3-N and Sol-P loadings with the statistical equations were all slightly less than

380   the values produced with SWAT for the different BMPs (Table 6). Mean NO_3-N loadings

381   were greater for BMP 1 (44 kg ha^{-1}) than those for BMPs 2 and 3 with both giving 39 kg

382   ha^{-1} (Table 6), due to increased infiltration with FDT. Mean Sol-P loading (0.8 kg ha^{-1})

383   was less for BMP 3 than for BMP 2 (0.89 kg ha^{-1}), whereas much greater than for BMP 1

384   (0.43 kg ha^{-1}). Although contour tillage can help reduce sediment loading by modifying

385   micro-topography and reducing erosion runoff (the reason we set USLE_P < 1), Sol-P

386   transported with surface runoff increased due to reduced residue cover protecting the soil

387   surface during winter and during the snowmelt season. When FDT was implemented with

388   tillage, however, less surface runoff was generated due to increased infiltration leading to

389   a reduction in Sol-P loading. Mean NO_3-N and Sol-P loadings for the FORT land

390   grouping (10 vs. 0.06 kg ha^{-1}) were much less than those of the CRGP land grouping, 39

391   vs. 0.8 kg ha^{-1} (Table 6).

392 **Table 6** Comparisons of simulations of statistical models and outputs from SWAT for different land use groups and BMPs based on

393 mean and standard deviation for the entire simulation period (1992-2011).

| Variable | Index | No-BMP | | | | | | Tillage | | FDT + Tillage | |
| | | CRGP | | NOCR | | FORT | | CRGP | | CRGP | |
		SWAT	Fitted	SWAT	Fitted	SWAT	Fitted	SWAT	Fitted	SWAT	Fitted
Discharge (mm)	Mean	→	→	636	636	←	←	619	619	628	628
	SD	→	→	144	133	←	←	140	132	151	143
	R^2	→	→	0.86 (Dis1)		←	←	0.88 (Dis2)		0.90 (Dis3)	
Sediment (t ha^{-1})	Mean	12.84	10.78	1.80	1.71	0.10	0.09	9.40	8.31	0.84	0.89
	SD	11.86	9.44	1.94	1.95	0.14	0.16	8.28	7.38	2.72	1.18
	R^2	0.48 (Sed1_1)		0.71 (Sed1_2)		0.57 (Sed1_3)		0.56 (Sed2)		—	
NO$_3$-N (kg ha^{-1})	Mean	43	39	24	—	10	10	43	39	47	44
	SD	24	14	16	—	6	3	24	14	29	21
	R^2	0.40 (N1_1)		—		0.33 (N1_3)		0.39 (N2)		0.59 (N3)	
Sol-P (kg ha^{-1})	Mean	0.88	0.80	0.61	—	0.08	0.06	0.98	0.89	0.49	0.43
	SD	0.49	0.32	0.46	—	0.06	0.03	0.59	0.38	0.33	0.23
	R^2	0.47 (P1_1)		—		0.38 (P1_3)		0.48 (P2)		0.52 (P3)	

394 Note: CRGP refers to crop groups including AGRL, GRAN, and GRAS; the statistics for discharge in no-BMP scenario are

395 based on CRGP, NOCR, and FORT.

396   **3.2   LBAT Assessment**

397   **3.2.1   Impact of Grid Cell Size on LBAT Simulation**

398   Simulations of water quantity and quality by LBAT with different grid-cell sizes (i.e.,

399   25, 50, 100, 200, and 400 m) for BBW are shown in Fig. 3. Statistical tests indicated that

400   grid-cell size had a significant effect on sediment loading (p-value < 0.01), with no effect

401   observed for discharge and NO_3-N and Sol-P loadings (p-values > 0.99). Increasing cell

402   size (i.e., slope length) increased sediment loading. However, the mean slope gradient

403   was reduced. As a result, the mean sediment loadings were correlated non-linearly with

404   cell size as shown in Fig. 4. The highest mean sediment loading was found with a cell

405   size of 100 m (5.86 t ha^{-1}), whereas the lowest was found to occur with a cell size of 25

406   and 400 m (3.37 t ha^{-1}). The LBAT with a cell size of 25 and 400 m was able to generate

407   sediment loadings consistent with field measurements. Considering computational

408   efficiency, we chose a grid-cell size of 400 m as the basic LBAT-simulation unit for

409   LRW.

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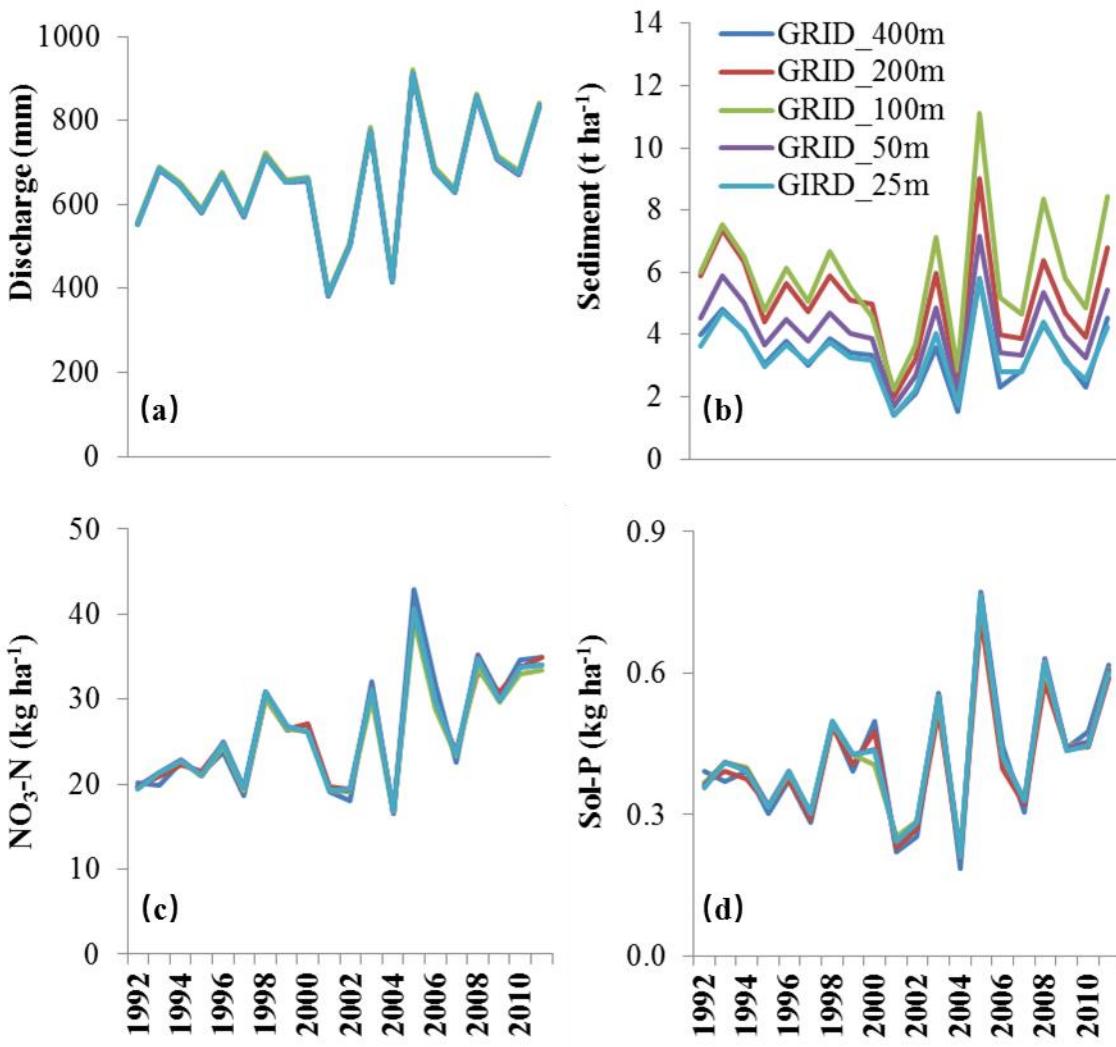
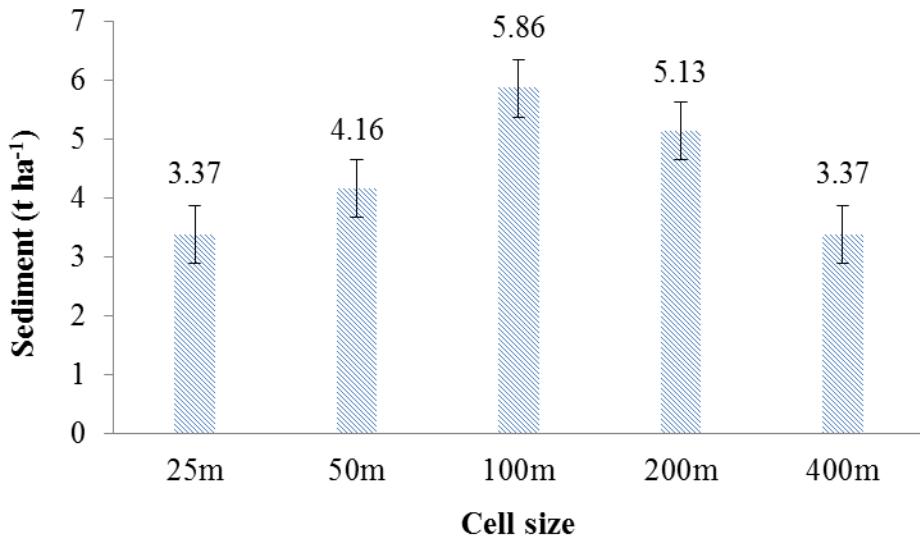


Fig. 3 LBAT-produced simulations of annual stream discharge and sediment, NO_3-N, and

Sol-P loadings determined for different DEM grid-cell sizes (i.e., 25, 50, 100, 200, and

400 m).

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415   **Fig. 4** Impact of grid-cell size on LBAT-simulation of sediment loading. Mean annual

416   sediment loadings and standard errors (vertical bars) from 1992 to 2011 are indicated.

3.2.2 LBAT vs. SWAT in LRW

Simulations of water quantity and quality with LBAT and the uncalibrated and calibrated versions of SWAT are compared with field measurements for LRW (Fig. 5). Model assessments for different simulation periods (depending on measurement availability) are shown in Table 7. It is worth noting that, to eliminate unrealistic results, USLE_LS was constrained in Sed1_2 to the NOCR land use group:

$$USLE\_LS = \begin{cases} Eq.\,6\text{-}1 & USLE\_LS \leq 1.28 \\ 1.28 & USLE\_LS > 1.28 \end{cases} \tag{8}$$

where 1.28 is the maximum USLE_LS for BBW.

In general, the two versions of SWAT and LBAT slightly underestimated annual stream discharge, capturing its variation reasonably well ($R^2 > 0.54$; Fig. 5a). The uncalibrated and calibrated versions of SWAT had the least and largest absolute values of Re (Re = -2 and -9), whereas LBAT Re = -6 (Table 7). The uncalibrated version of SWAT severely overestimated annual sediment and NO_3-N loading (Re = 212 and 87, respectively; Figs. 5b and c), whereas the calibrated version of SWAT and LBAT underestimated sediment loading (Re = -32 and -52, respectively) and overestimated NO_3-N loading (Re = 22 and 27, respectively; Table 7). In general, the calibrated version of SWAT and LBAT captured the variation in annual NO_3-N loadings reasonably well ($R^2 > 0.35$; Fig. 5c). However, the two versions of SWAT and LBAT failed to capture the variation in annual sediment and Sol-P loadings (low R^2; Figs. 5b and d). The LBAT had the smallest absolute value of Re (i.e., Re = -16), while the uncalibrated and calibrated versions of SWAT had larger values (Re = -59 and -55, respectively). These results suggested that the LBAT and the calibrated version of SWAT performed fairly equivalently in simulating annual stream flow and sediment and NO_3-N loadings, with

LBAT performing slightly better for annual Sol-P loading. LBAT performed noticeably

better than the uncalibrated version of SWAT, especially for annual sediment and NO$_3$-N

loadings. Poor performance for both versions of SWAT and LBAT on simulation of

annual sediment and Sol-P loadings in LRW might attribute to lack of detailed

management practice and fertilizer application information from agricultural lands. We

only had one-year data for LRW and made assumptions about rotation and management

practices for other years based on information from BBW, which could introduce major

input uncertainty.

Table 7 Statistical assessments of LBAT and SWAT for annual stream discharge and

sediment, NO$_3$-N, and Sol-P loadings at the outlet of LRW for different simulation

periods

Period	Variable	Index	Measurement	SWAT -Uncalibrated	SWAT -Calibrated	LBAT
01-07	Discharge (mm)	Mean	704	691	638	664
		Re (%)	—	-2	-9	-6
		R^2	—	0.63	0.69	0.54
01-10	Sediment (t ha^{-1})	Mean	0.95	2.95	0.65	0.45
		Re (%)	—	212	-32	-52
		R^2	—	0.01	0.01	0.04
03-10	NO$_3$-N (kg ha^{-1})	Mean	12	22	14	15
		Re (%)	—	87	22	27
		R^2	—	0.59	0.45	0.35
03-10	Sol-P (kg ha^{-1})	Mean	0.31	0.13	0.14	0.26
		Re (%)	—	-59	-55	-16
		R^2	—	0.02	0.11	0.01

463   Since LBAT is based on decision rules (statistical equations in this study) which

464 were derived from SWAT simulations for BBW, its usage should be constrained to areas

465 with soil, landscape, and land use characteristics similar to BBW. Input characteristics

466 exceeding the range of SWAT data could lead to large errors in predictions. LBAT is

467 flexible in its structure, and with thoughtful development of decision rules, it can be

468 applied to diverse environments.

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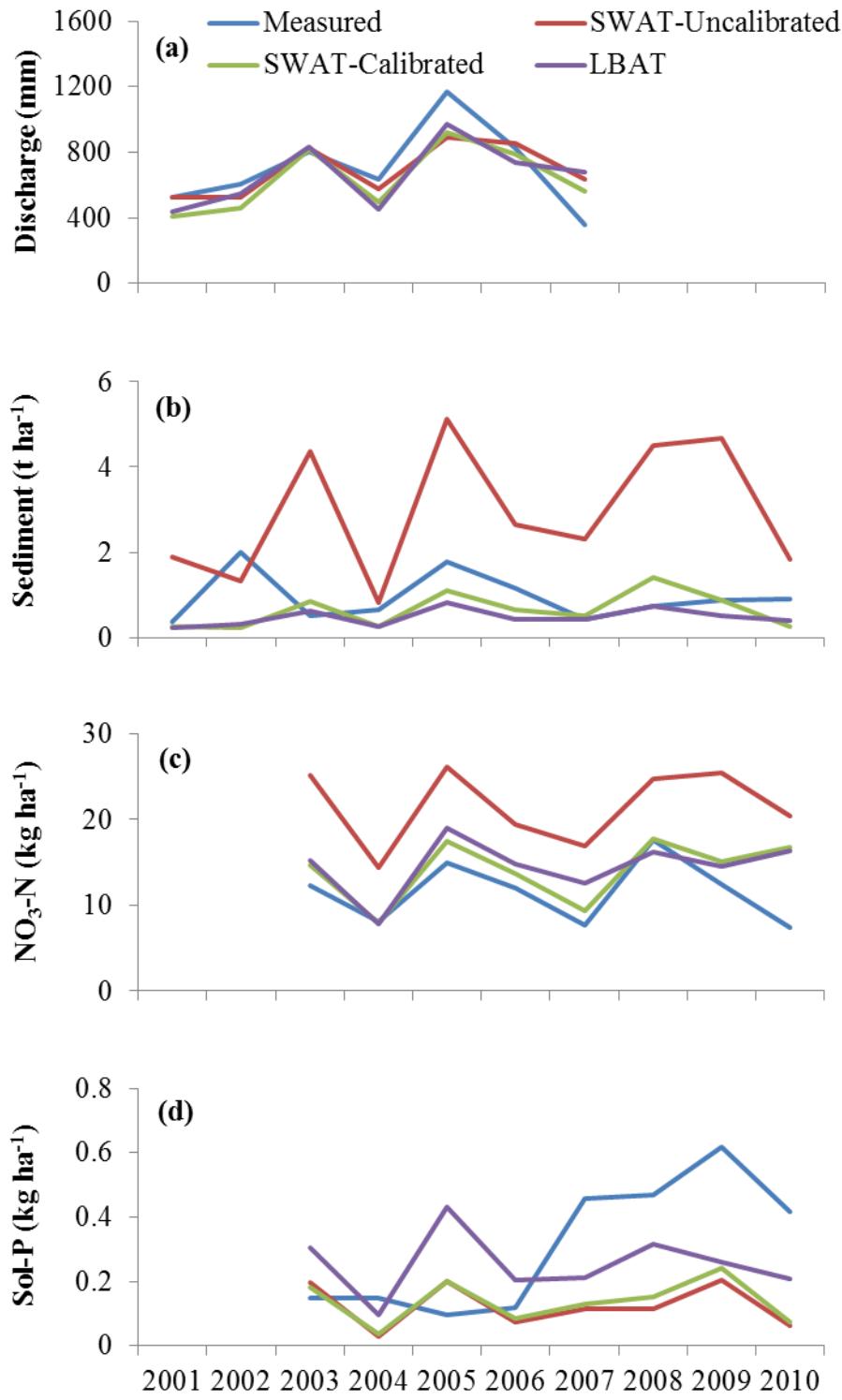
Fig. 5 Simulations of annual stream discharge and sediment, NO_3-N, and Sol-P loadings

with LBAT and SWAT compared with field measurements at the outlet of LRW.

472

3.2.3 FDT Assessment in LRW

474   Mean annual water quantity and quality simulated with LBAT for agricultural lands of

475   LRW are shown in Table 8. The mean annual discharge for the baseline scenario was 626

476   mm greater than that for the six FDT scenarios (Table 8). When all agricultural lands

477   were protected (S6), there was a 2% reduction in discharge (equivalent to 11 mm; Table

478   8). With the steepest areas protected (accounting for 10% of the total land base; S1), the

479   mean annual sediment loading was reduced by as much as 43% (equivalent to 4.5 t ha^{-1};

480   Table 8) and by as much as 81% (i.e., 8.57 t ha^{-1}) with all agricultural lands protected (S6;

481   Table 8). Mean annual Sol-P loading was reduced by 51% (equivalent to 0.47 kg ha^{-1};

482   Table 8). In contrast, increased usage of FDT tended to increase the mean annual loading

483   of NO_3-N, by about 6% when used across all agricultural lands (equivalent to 1.73 kg ha^{-}

484   ^{1}).

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494 **Table 8** Impact of FDT on mean annual discharge and sediment, NO_3-N, and Sol-P

495 loadings simulated with LBAT under different FDT, provided in Table 3.

Variable	Index	Baseline	S1	S2	S3	S4	S5	S6
Discharge (mm)	Mean	626	625	623	622	619	616	615
	MD	—	-1	-2	-4	-7	-10	-11
	PRD (%)	—	0	0	-1	-1	-2	-2
Sediment ($t\ ha^{-1}$)	Mean	10.54	6.04	4.94	4.02	3.04	2.26	1.97
	MD	—	-4.50	-5.60	-6.52	-7.50	-8.28	-8.57
	PRD (%)	—	-43	-53	-62	-71	-79	-81
NO_3-N ($kg\ ha^{-1}$)	Mean	29.70	29.86	30.02	30.34	30.82	31.22	31.42
	MD	—	0.16	0.32	0.64	1.13	1.52	1.73
	PRD (%)	—	1	1	2	4	5	6
Sol-P ($kg\ ha^{-1}$)	Mean	0.94	0.89	0.83	0.76	0.65	0.52	0.46
	MD	—	-0.05	-0.11	-0.17	-0.28	-0.42	-0.47
	PRD (%)	—	-5	-11	-19	-30	-45	-51

496

497   Percentage change (based on PRD) of water quantity and quality were plotted against

498 percentage area of FDT for potato and barley in Fig. 6. Increasing the usage of FDT

499 helped to reduce discharge and sediment and Sol-P loadings for both crop types (Figs. 6a,

500 b, and c). It is worth noting that sediment loading decreased with increasing usage of

501 FDT (Fig. 6b). An opposite trend was observed for potato and barley with respect to the

502 impact of FDT on NO_3-N loading. With the increased usage of FDT, NO_3-N loadings

503 increased linearly for potato, while it decreased for barley. The increased for potato was

504 nearly twice as much as the reduction for barley (Fig. 6d). Seemingly the interaction

505 between barley and FDT had positive impacts on nitrate retention in soils, whereas the

506 interaction between potato and FDT had an opposite effect.

507   These results are consistent with the results from previous studies (Yang et al.,

508 2012;Yang et al., 2010), which used SWAT to assess the impact of FDT on water

509 quantity and quality within BBW. When using SWAT, greater efforts are needed to

510   prepare basic inputs, such as daily weather records, to proceed with its calibration and

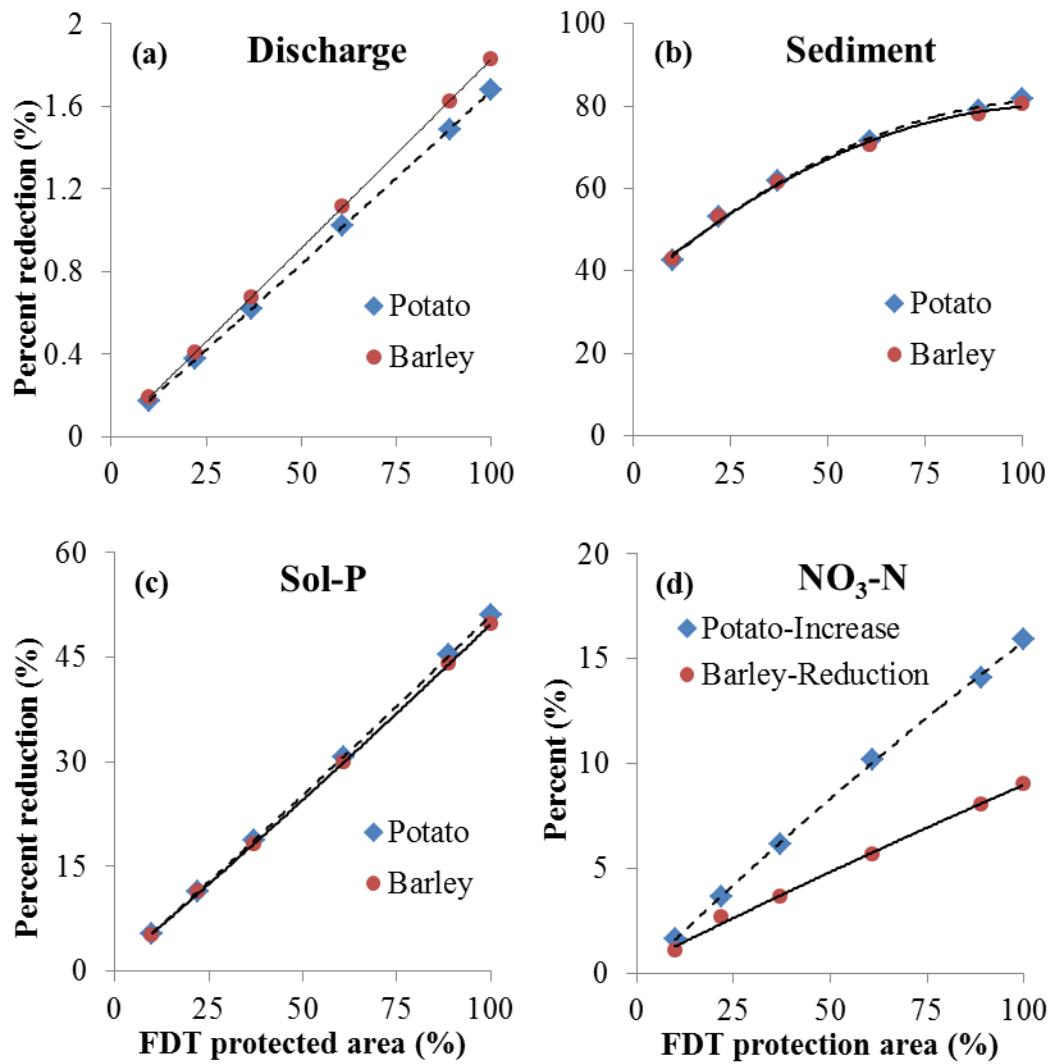
511   validation, involving complex scenario setup and analysis. For every new watershed,

512   SWAT needs dedicated effort and time for its setup. LBAT, in contrast, can be used for

513   multiple watersheds as long as they have similar environmental conditions. Scenario

514   analysis can be directly conducted with different combinations of land use and BMPs

515   using fewer inputs than what is required by SWAT. Also, once developed, LBAT does

516   not require additional calibration.

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518   **Fig. 6** Percentage change in discharge and sediment, NO_3-N, and Sol-P loadings as a

519   function of % area, where FDT's were used.

4. Conclusion

The present study addresses the development of a decision support tool to assess the impact of land use change and BMPs on water quantity and quality for ungauged watersheds. An enhanced version of SWAT was calibrated and validated for an small experimental watershed. Multiple regression analyses were used to develop statistical equations based on simulations from SWAT. In total, three discharge and five sediment, NO_3-N, and Sol-P loading models were developed for different combinations of land use groups and BMP scenarios. Only four common predictors (i.e., annual precipitation, annual mean air temperature, mean saturated hydraulic conductivity of soil, and land use groups) and three unique predictors (LS-factor and annual nitrogen and phosphorus application rates for sediment, NO_3-N, and Sol-P loading models, respectively) are required.

With the aid of ArcGIS, statistical equations were integrated into the decision support tool, i.e., the land use and BMPs assessment tool (LBAT), whose basic simulation units are the DEM-grid cell. The LBAT was used to simulate annual water flow and sediment and nutrient loadings at the outlet of a larger watershed, i.e., Little River Watershed (LRW). These simulations were compared with those of SWAT. Results indicated that LBAT and the calibrated version of SWAT performed equivalently with respect to annual stream discharge and sediment and NO_3-N loadings. LBAT performed slightly better, when Sol-P loading was considered. Compared with the uncalibrated version of SWAT, LBAT performed better. The impact of FDT on water quantity and quality was evaluated with LBAT for LRW; its results were consistent with the results generated with SWAT for the same region in previous studies. LBAT has fewer input requirements than SWAT,

543 and can be applied to multiple watersheds without additional calibration. Also, scenario

544 analyses can be directly conducted with LBAT without complex setup procedures. We

545 recommend using LBAT for economic analysis and management decision making for

546 watersheds with similar environmental conditions of New Brunswick. The LBAT

547 developed in this study may not be directly applied to other regions; however, the

548 approach in developing LBAT can be applied to other regions of the world because of its

549 flexible structure.

550

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 Appendix A

 Table A1 Coefficient values for the three discharge models.

Model variable	Estimate	Std. Error	t-value	*p*-value
Dis1				
Intercept	-1565	24.04	-65.089	<0.001
PCP	1.933	0.02176	88.837	<0.001
TMP	282.7	6.091	46.402	<0.001
SOL_K	0.06338	0.00992	6.389	<0.001
FORT	30.79	14.16	2.175	0.030
NOCR	162.2	14.51	11.181	<0.001
PCP:TMP	-0.2488	0.005487	-45.352	<0.001
PCP:FORT	0.04684	0.01191	3.934	<0.001
PCP:NOCR	-0.0535	0.01224	-4.37	<0.001
TMP:FORT	9.723	1.684	5.775	<0.001
TMP:NOCR	4.506	1.731	2.603	0.009
SOL_K:FORT	-0.3769	0.03403	-11.076	<0.001
SOL_K:NOCR	-0.2959	0.032	-9.248	<0.001
Dis2				
Intercept	-1633	27.29	-59.84	<0.001
PCP	1.995	0.02472	80.69	<0.001
TMP	302.2	6.87	43.98	<0.001
SOL_K	0.08696	0.01167	7.45	<0.001
PCP:TMP	-0.2662	0.006199	-42.94	<0.001
Dis3				
Intercept	-1666	36.58	-45.54	<0.001
PCP	2.007	0.03305	60.713	<0.001
TMP	298	9.351	31.865	<0.001
SOL_K	0.09353	0.01573	5.946	<0.001
PCP:TMP	-0.2606	0.008406	-31.004	<0.001

575 **Table A2** Coefficient values for the four sediment loading models.

Model variable	Estimate	Std. Error	t-value	*p*-value
Sed1_1				
Intercept	0.2749	0.06125	4.488	<0.001
USLE_LS	0.1201	0.02224	54.018	<0.001
PCP	0.000788	5.54E-05	14.218	<0.001
TMP	0.1117	0.01528	7.307	<0.001
SOL_K	0.000568	0.00022	2.585	0.010
GRAS	-0.0353	0.00881	-4.007	<0.001
USLE_LS:SOL_K	-0.00014	4.69E-05	-3.045	0.002
USLE_LS:GRAS	-0.02623	0.006826	-3.842	<0.001
PCP:TMP	-0.00011	1.38E-05	-7.967	<0.001
PCP:SOL_K	-4.6E-07	1.91E-07	-2.406	0.016
Sed1_2				
Intercept	0.8575	0.008826	97.15	<0.001
PCP	0.000123	7.82E-06	15.67	<0.001
PCP:USLE_LS	0.000209	5.02E-06	41.65	<0.001
Sed1_3				
(Intercept)	0.3992	0.02267	17.613	<0.001
USLE_LS	0.07935	0.01967	4.034	<0.001
PCP	0.000204	1.96E-05	10.371	<0.001
SOL_K	0.000545	5.71E-05	9.534	<0.001
USLE_LS:PCP	4.94E-05	1.71E-05	2.9	0.004
USLE_LS:SOL_K	-0.00067	4.89E-05	-13.718	<0.001
Sed2				
Intercept	0.2591	0.05228	4.956	<0.001
USLE_LS	0.12	0.001898	63.218	<0.001
PCP	0.000767	4.73E-05	16.212	<0.001
TMP	0.1162	0.01304	8.907	<0.001
SOL_K	0.000746	0.000188	3.981	<0.001
GRAS	-0.06937	0.01648	-4.211	<0.001
USLE_LS:SOL_K	-0.00013	4E-05	-3.137	0.002
USLE_LS:GRAS	-0.02662	0.005829	-4.567	<0.001
PCP:TMP	-0.00011	1.18E-05	-9.522	<0.001
PCP:SOL_K	-6.3E-07	1.63E-07	-3.846	<0.001
TMP:GRAS	0.007415	0.003664	2.024	0.043

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579 **Table A3** Coefficient values for the four NO₃-N loading models corresponding to land

580 use and BMPs described in Table 4.

Model variable	Estimate	Std. Error	t-value	*p*-value
N1_1				
Intercept	1.44	0.1753	8.213	<0.001
N_APP	-0.00862	0.000699	-12.325	<0.001
PCP	0.000543	0.00016	3.4	<0.001
TMP	0.1363	0.03357	4.059	<0.001
SOL_K	-0.00344	9.78E-05	-35.163	<0.001
GRAN	-1.117	0.1021	-10.937	<0.001
GRAS	-1.97	0.1562	-12.611	<0.001
N_APP:PCP	5.31E-06	6.45E-07	8.233	<0.001
N_APP:TMP	0.000963	7.45E-05	12.929	<0.001
N_APP:SOL_K	9.6E-06	6.4E-07	15.024	<0.001
PCP:GRAN	0.000677	9.38E-05	7.215	<0.001
PCP:GRAS	0.001029	0.000143	7.201	<0.001
PCP:TMP	-0.00025	2.64E-05	-9.467	<0.001
TMP:GRAN	0.1	0.01134	8.817	<0.001
TMP:GRAS	0.2132	0.01651	12.912	<0.001
N1_3				
Intercept	-1.411	0.3087	-4.573	<0.001
PCP	0.001875	0.000279	6.710	<0.001
TMP	0.4437	0.07831	5.666	<0.001
SOL_K	-0.00104	0.000116	-8.979	<0.001
PCP:TMP	-0.00032	7.06E-05	-4.484	<0.001
N2				
Intercept	1.429	0.1757	8.134	<0.001
N_APP	-0.00858	0.000701	-12.233	<0.001
PCP	0.000548	0.00016	3.425	<0.001
TMP	0.1376	0.03365	4.089	<0.001
SOL_K	-0.00345	9.8E-05	-35.223	<0.001
GRAN	-1.11	0.1023	-10.849	<0.001
GRAS	-1.962	0.1566	-12.526	<0.001
N_APP:PCP	5.3E-06	6.47E-07	8.187	<0.001
N_APP:TMP	0.000957	7.46E-05	12.82	<0.001
N_APP:SOL_K	9.65E-06	6.4E-07	15.067	<0.001
PCP:GRAN	0.000674	9.41E-05	7.167	<0.001
PCP:GRAS	0.001026	0.000143	7.162	<0.001
PCP:TMP	-0.00025	2.64E-05	-9.456	<0.001
TMP:GRAN	0.09934	0.01137	8.738	<0.001
TMP:GRAS	0.2122	0.01655	12.821	<0.001

N3				
Intercept	-0.3595	0.1718	-2.092	0.037
N_APP	-0.00131	0.000435	-3.011	0.003
PCP	0.001621	0.00015	10.806	<0.001
TMP	0.3977	0.03857	10.312	<0.001
SOL_K	-0.00386	0.000505	-7.641	<0.001
GRAN	-0.2133	0.07504	-2.842	0.005
N_APP:PCP	1.65E-06	3.59E-07	4.61	<0.001
N_APP:TMP	0.000281	4.74E-05	5.939	<0.001
N_APP:GRAN	0.000716	0.000292	2.453	0.014
PCP:TMP	-0.00035	3.32E-05	-10.506	<0.001
PCP:SOL_K	1.21E-06	4.36E-07	2.781	0.005
PCP:GRAN	0.000267	5.82E-05	4.577	<0.001
TMP:GRAN	-0.04685	0.008004	-5.853	<0.001

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Table A4 Coefficient values for four Sol-P models.

Model variable	Estimate	Std. Error	t-value	*p*-value
P1_1				
Intercept	-3.711	0.1306	-28.416	<0.001
P_APP	0.002341	0.000623	3.757	<0.001
PCP	0.003195	0.000117	27.286	<0.001
TMP	0.5542	0.03197	17.337	<0.001
SOL_K	0.00298	0.000472	6.305	<0.001
GRAS	-0.4321	0.0382	-11.312	<0.001
P_APP:PCP	-2.4E-06	5.2E-07	-4.64	<0.001
P_APP:TMP	0.000829	7.7E-05	10.797	<0.001
PCP:TMP	-0.00052	2.9E-05	-18.297	<0.001
PCP:SOL_K	-1.2E-06	3. 97E-07	-3.095	0.002
TMP:SOL_K	-0.00026	5.7E-05	-4.526	<0.001
TMP:GRAS	0.03787	0.00941	4.024	<0.001
P1_3				
Intercept	-4.43817	0.589848	-7.512	<0.001
PCP	0.002509	0.000534	4.701	<0.001
TMP	0.417306	0.1496445	2.789	0.005
SOL_K	0.001247	0.000222	5.622	<0.001
PCP:TMP	-0.0003	0.000135	-2.253	0.024
P2				
Intercept	-3.667	0.1357	-27.017	<0.001
P_APP	0.003461	0.000663	5.218	<0.001
PCP	0.003017	0.000122	24.783	<0.001
TMP	0.5149	0.03304	15.584	<0.001
SOL_K	0.003531	0.000488	7.233	<0.001
GRAS	-0.2039	0.09001	-2.265	0.024
P_APP:PCP	-2.4E-06	5.54E-07	-4.305	<0.001
P_APP:TMP	0.000432	7.93E-05	5.445	<0.001
P_APP:GRAS	-0.03304	0.007019	-4.707	<0.001
PCP:TMP	-0.00044	2.95E-05	-14.952	<0.001
PCP:SOL_K	-1.4E-06	4.1E-07	-3.446	<0.001
PCP:GRAS	-0.00025	7.66E-05	-3.25	0.001
TMP:SOL_K	-0.00025	5.87E-05	-4.184	<0.001
TMP:GRAS	0.05117	0.009839	5.201	<0.001
P3				
Intercept	-2.817	0.2548	-11.054	<0.001
P_APP	-0.01363	0.001854	-7.352	<0.001
PCP	0.002778	0.000178	15.609	<0.001
TMP	0.1406	0.06523	2.155	0.031
SOL_K	0.00651	0.000702	9.279	<0.001

GRAN	-0.9386	0.1378	-6.812	<0.001
GRAS	-0.9931	0.1813	-5.478	<0.001
P_APP:TMP	0.003562	0.000491	7.252	<0.001
P_APP:GRAN	0.007736	0.002179	3.549	<0.001
P_APP:GRAS	-0.05489	0.01295	-4.24	<0.001
PCP:TMP	-0.0003	4.42E-05	-6.763	<0.001
PCP:SOL_K	-3.7E-06	5.78E-07	-6.359	<0.001
PCP:GRAN	0.000112	5.1E-05	2.192	0.028
PCP:GRAS	-0.00019	0.000109	-1.74	0.082
TMP:SOL_K	-0.00021	8.8E-05	-2.4	0.016
TMP:GRAN	0.1798	0.03332	5.397	<0.001
TMP:GRAS	0.247	0.03581	6.898	<0.001

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