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# Modeling effectiveness of agricultural BMPs to reduce sediment load and organophosphate pesticides in surface runoff

## Xuyang Zhang <sup>a,\*,\*\*</sup>, Minghua Zhang <sup>a,b,\*\*</sup>

<sup>a</sup> California Department of Pesticide Regulation, 1001 I Street, Sacramento, CA, 95616, United States

<sup>b</sup> Department of Land, Air & Water Resources, University of California, 1 Shields Ave, Davis, CA, 95616, United States

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#### ABSTRACT

Quantifying effectiveness of agricultural BMPs at the watershed scale is a challenging issue, requiring robust algorithms to simulate not only the agricultural production system but also pollutant transport and fate. This research addresses the challenge to simulate performances of BMPs in reducing organophosphates (OPs) runoff at the watershed scale. The SWAT model is calibrated and validated following a sensitivity analysis combining Latin Hypercube sampling and One-factor-At-a-Time simulation. The calibrated model is then applied in the Orestimba Creek Watershed to simulate BMPs including buffer strips, sediment ponds, vegetated ditches, use reduction, and their combinations. BMP simulation suggested that sediment ponds trap 54–85% of sediment from field runoff, but less than 10% of dissolved diazinon and chlorpyrifos. Use reduction can reduce pesticide load in a close-to-linear fashion. Effectiveness of vegetated ditches and buffers depends on their physical dimension and vegetated news, buffer strips and use reduction decreases diazinon and chlorpyrifos load by over 94%. This study has suggested that the SWAT model reasonably predicts BMP effectiveness at the watershed scale. Results will assist decision making in implementing BMPs to reduce pesticide loads in surface runoff.

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## 1. Introduction

Widespread use of pesticides in modern agriculture contributes to agricultural non-point source pollution (ANPSP) in rivers and streams across the world. In the Central Valley of California, one of the major agricultural production areas of the world, use of broad-spectrum organophosphate (OP) pesticides such as diazinon and chlorpyrifos has resulted in their frequent detection in surface waters (Moore et al., 2008). During the last decade, water samples taken from dormant-season orchard drainages in California's Central Valley have been found to be toxic to the freshwater aquatic invertebrate *Ceriodaphnia dubia* (de Vlaming et al., 2000). To minimize the potential risk from pesticide use, measures should be taken to prevent pesticides from being transported offsite and consequently polluting receiving waters. Agricultural best management practices (BMPs) have been recognized as one of the best solutions to mitigate ANPSP.

BMPs are structural or non-structural management practices that aim to reduce the impacts of sediment and agrochemicals on water quality. Some examples of BMPs include, but are not limited to, sediment

\*\* Correspondence to: X. Zhang and M. Zhang, California Department of Pesticide Regulation, 1001 I Street, Sacramento, CA, 95616, United States. Tel.: +1 916 445 3195; fax: +1 916 445 4405.

ponds, vegetated buffers, constructed wetlands, and vegetated ditches. These BMPs have been shown to be effective in removing agrochemicals and sediments from field runoff (Budd et al., 2009, Zhang et al., 2009; Bennett et al., 2009). Growers in California are required to implement BMPs to reduce ANPSP in their discharge water if pollutant exceedances are measured from their land. Regulatory enforcement of BMP implementation results in an urgent need for quantitative information on BMP effectiveness for agricultural runoff (Lee and Jones-Lee, 2002).

BMP effectiveness has been studied mainly through field experiments. but computer modeling has been increasingly used as a valuable alternative. While field experiments are costly and difficult to repeat, computer simulations can be run to test various implementation scenarios. In addition, watershed models simulate BMP effectiveness at larger scales, which cannot be feasibly achieved by field experiments. Modeling effectiveness of agricultural BMPs at the watershed scale is a challenging issue, requiring robust algorithms to simulate not only the agricultural production system, but also pollutant transport and fate. The Soil and Water Assessment Tools (SWAT) model is one of the very few models that meet such requirements. The model has proven to be an effective tool for evaluating BMP implementation, alternate land use, and other factors contributing to lower pollutant levels (Arabi et al., 2008; Gassman et al., 2007; Bracmort et al., 2006; Stewart et al., 2006; Chaplot et al., 2004; Whitall et al., 2004; Santhi et al., 2001). However, very few of the studies were conducted to evaluate the effects of BMPs on pesticide reduction. While modeling pesticide transport and fate is often more complicated than hydrological simulation (Holvoet et al., 2005), research

<sup>\*</sup> Corresponding author. Tel.: +1 916 445 3195; fax: +1 916 445 4405.

*E-mail addresses*: xzhang@cdpr.ca.gov (X. Zhang), mhzhang@ucdavis.edu (M. Zhang).

efforts are needed to fill this important knowledge gap in order to successfully implement BMPs for alleviating ANPSP. Luo and Zhang (2009) performed a preliminary analysis on the effects of BMPs in reducing pesticide runoff following a sensitivity analysis for pesticide transport using SWAT. However, the BMP scenarios were much simpler with individual BMPs implemented uniformly in a watershed. This research extends the work to include a comprehensive array of BMPs and their combination scenarios. Objectives of the study are to (1) simulate the effectiveness of each BMP scenario; (2) identify the most effective BMPs for reducing pesticide loads; and (3) provide the information necessary to assist decision making in implementing BMPs to reduce diazinon and chlorpyrifos pesticides in surface runoff.

## 2. Materials and methods

## 2.1. Watershed description

Orestimba Creek is a tributary of the San Joaquin River located in the western San Joaquin Valley (Fig. 1). The lower portion of Orestimba Creek (Lower Orestimba Creek) flows across a 146 km<sup>2</sup> stretch of irrigated agricultural land, which is planted with various crops including alfalfa, walnuts, almonds, irrigated pasture, dry beans, tomatoes and corn. During 2000 and 2006, an annual total of 2870 kg of chlorpyrifos and 402 kg of diazinon were applied during both the irrigation and rainy seasons in the watershed (CDPR, 2008). Both pesticides have been frequently detected in Lower Orestimba Creek. As a result, the creek has been designated as an impaired water body on the 303 (d) list (California State Water Resource Control Board, 2002). A USGS gage station (Orestimba Creek at River Road near Crows Landing, California, OCCL) is located near the confluence of Orestimba Creek with the San Joaquin River. The station receives water

discharge from rainfall runoff during the rainy season and irrigation return flow during the rest of the year. Stream flow and water quality data collected since the early 1990s at this station allow model calibration for the watershed.

#### 2.2. Data sources

## 2.2.1. Spatial and temporal model input data

Model inputs, such as landscape and weather conditions, were compiled using databases from various agencies. Data for landscape descriptions, including elevation, land use, and stream network were obtained from the BASINS database, in which the SWAT model is integrated as a sub-model (USEPA, 2007). Retrieved data included 1:250,000 scale quadrangles of land use/land cover data, USGS 30 m resolution National Elevation Dataset (NED), and 1:100,000 scale National Hydrography Dataset (NHD). Contemporary cropland and irrigation areas in the Orestimba Creek watershed were defined based on the land-use survey database developed by the California Department of Water Resources in 2004 for Stanislaus and Merced counties. Soil properties were extracted from the 1:24,000 scale Soil Survey Geographic (SSURGO) database (USDA, 2009) based on soil surveys conducted in the study area during the 1990s. Daily meteorological data, including precipitation, solar radiation, minimum/maximum temperatures, relative humidity, and wind speed, were retrieved from the California Irrigation Management Information System (CIMIS) for the station at Patterson, CA (Fig. 1).

## 2.2.2. Monitoring data

Measured data of stream flow, sediment load and pesticide concentration were obtained from the National Water Information System (NWIS) maintained by USGS for the two monitoring sites (OCN, USGS



Fig. 1. Location of the Orestimba Creek Watershed.

11274500; OCCL, USGS 11274537). Monthly average stream flow rate was aggregated from daily data. In cases where data did not exist for a given month, the long-term monthly average was applied. Sediment concentrations were usually available at a monthly or biweekly interval. Instantaneous sediment load was calculated as the product of measured sediment concentration and stream flow and then averaged as monthly data. For the period 2000–2006, a total of 190 water samples for diazinon and chlorpyrifos concentrations were available at the watershed outlet OCCL. Pesticide load was calculated by multiplying concentration and stream flow on each day. Monthly load was estimated from average daily loads available in that month. Since the upper portion of Orestimba Creek is mainly covered by forest and rangeland, it was assumed that there was no pesticide input from that portion of the sub-basin.

#### 2.2.3. Pesticide use information

Pesticide use information from 2000 to 2006 was obtained from the PUR database maintained by the California Department of Pesticide Regulation (CDPR, 2008). The database records pesticide use information for every application in production agriculture, industrial and landscape maintenance. The database includes information on the amount of pesticide product used, amount of active ingredient used, application date, the planted area and the treated area. Locations of applications for production agriculture are recorded at the section level (approximately one square mile) of the Public Land Survey System. Use amounts of pesticide active ingredients were summarized for each crop, and distributed into the agricultural hydrological response units (HRUs) according to crop land use.

## 2.2.4. Pesticide physicochemical properties

Physicochemical properties of diazinon and chlorpyrifos were obtained from the Pesticide Chemical Database developed by California Department of Pesticide Regulation, which provides chemical property values based on a peer-review process evaluating the validity of the individual studies (CDPR, 2009) (Table 1). The key parameters include soil adsorption coefficient (SKoc), degradation half-lives in soil, foliage and water, solubility, Henry's law constant and various degradation coefficients (Table 1). Mass transport coefficients, such as settling velocity, resuspension velocity, mixing velocity, and burial velocity were set as their default values suggested by the SWAT model (Neitsch et al., 2005).

## 2.3. Model initialization

This study uses the ArcSWAT modeling package, which runs the 2005 version of the SWAT model within the ESRI ArcGIS 9.3 environment (Di Luzio et al., 2004; Winchell et al., 2007). The interface facilitates the preprocessing of the GIS data of elevation, soil, land use, and weather as basic model inputs. The watershed was delineated into two sub-basins and 12 hydrological response units, which are unique combinations of land use, soil type and slope. Model simulation begins in 2000 since flow rates in Orestimba Creek changed dramatically after 1999. Measured daily average

#### Table 1

Physicochemical properties of diazinon and chlorpyrifos.

Description	SWAT parameter	Diazinon	Chlorpyrifos
Soil adsorption coefficient	SKOC	1000	6070
Wash-off fraction	WOP	0.9	0.65
Half-life on foliage (day)	HLIFE_F	4.0	3.3
Half-life in soil (day)	HLIFE_S	30	90
Solubility (mg/l)	WSOL	60	0.4
Molecular weight (g/mol)	MW	304.4	350.6
Henry's law constant	HENRY	$3.0 \times 10^{-5}$	$3.0 \times 10^{-4}$
Hydrolysis coefficient (day <sup>-1</sup> )	CHPST_REQ	0.005	0.01
Degradation coefficient in sediment (day <sup>-1</sup> )	SEDPST_REA	0.043	0.005

flow during 1990–1999 were 0.764 and 1.673 m<sup>3</sup>/s for the ORN and ORCL stations, respectively, which are located in the upper and lower stream. In 2000–2006, flow rates for sites ORN and ORCL changed to 0.318 and 0.869 m<sup>3</sup>/s, respectively. The simulation period was between 2000 and 2006 with the first two years used for model initialization. The purpose of model initialization was to allow state variables to be calculated from forcing variables rather than user-defined initial values, which might not reflect actual temporal variations. Model calibration was performed on data from 2003 to 2005 and data from 2006 was used for model validation.

Multiple options for runoff generation and evapotranspiration estimates were available in the SWAT model. Preliminary analysis showed that the best combination of runoff generation and evapotranspiration was the curve number and Priestley–Taylor method, respectively (Luo et al., 2008). Pesticide simulation has been found to be more reliable when performed on a monthly basis rather than daily due to possible time shifts in precipitation, agricultural activity and measurements for flow and pesticide concentration. Therefore, model predictions were reported and evaluated on a monthly and annual basis (Luo et al., 2008). Flow and sediment simulation were calibrated at the watershed outlet with data from the ORCL gage station (USGS site# 11274538).

#### 2.4. LH-OAT sensitivity analysis

The LH-OAT sensitivity analysis is a hybrid approach combining the Latin-Hypercube simulation (LH) with the One-Factor-At-a-Time (OAT) sampling methods. The LH simulation concept is based on Monte Carlo simulation but uses a stratified sampling approach to reduce the number of simulations. The approach, therefore, inherits the robustness of the Monte Carlo simulation, while requiring less simulation runs and computational resources (van Griensven et al., 2006). It subdivides the distribution of each parameter into N ranges, and randomly samples values from each range. Only one sample within each range was extracted within each run. The model runs N times with the random combination of parameters. LH is commonly applied in hydrological modeling due to its robustness and high efficiency (Weijers and Vanrolleghem, 1997; Vandenberghe et al., 2001).

The Morris OAT is a sensitivity analysis technique that integrates both local and global sensitivity (Morris, 1991). Only one parameter is changed in each run so that the variation of model output can be unambiguously attributed to the parameter changed (Morris, 1991). For each parameter, local sensitivities are computed at different points of the parameter range, and then the global effect is obtained by taking their average. In this way, local sensitivities are integrated into a global sensitivity. The advantage of the OAT method is its independence on the predefined assumptions, such as monotonicity of outputs with respect to inputs, few inputs having important effects and adequacy of low-order polynomial models as an approximation to the computation model (Morris, 1991).

The LH-OAT method takes the LH samples as the initial point. Around each LH point j, a partial effects  $S_{i,j}$  for each parameter is calculated using Eq. (1) (van Griensven et al., 2006).

$$S_{ij} = \frac{\left| 100 \times \left\{ \frac{M(\varphi_1, ..., \varphi_i \times (1+f), ..., \varphi_p) - M(\varphi_1, ..., \varphi_i, ..., \varphi_p)}{\left[ M(\varphi_1, ..., \varphi_i \times (1+f), ..., \varphi_p) + M(\varphi_1, ..., \varphi_i, ..., \varphi_p) \right] / 2} \right\}}{f}$$
(1)

where  $S_{i,j}$  is a partial effect for parameter  $\varphi_i$ , M ( $\cdot$ ) refers to the model functions, and f is the fraction by which the parameter  $\varphi_i$  is changed. The method operates in loops, with only one input parameter being modified in each loop. A final effect is calculated by averaging the partial effects of each loop for all LH points. Thus, the LH-OAT method combines the robustness of the LH sampling that ensures the full range of all parameters being sampled, with the precision of the OAT design assuring that changes in model output can be unambiguously

attributed to the change of parameter  $\varphi_i$ . Sensitivities of parameters are ranked by final effects with the largest effect being given rank 1 and the smallest effect being given a rank equal to the total number of parameters selected for analysis.

Sensitivity analysis was performed for 30 parameters identified in the literature to have a potential influence on flow, sediment yield and pesticide loads. Ranges of the parameters were based on the SWAT manual (Neitsch et al., 2005). LH samples were taken by dividing parameter ranges into 10 intervals, each with one starting point for parameter adjustment. During each loop of the model run, one of the starting points was then changed incrementally by a fraction of 0.05.

Sensitivity analysis indicated that pesticide transport in Orestimba Creek was determined by surface runoff, and that pesticide fate was highly dependent on the pesticide's physicochemical properties and plant morphology. Table 2 shows the most sensitive parameters and their ranks from the LH-OAT analysis. The most sensitive parameters for surface flow include SCS curve number (CN2), soil evaporation compensation factor (ESCO), fraction of ground water recharge to deep aquifer (RCHR\_DP), soil depth (SOL\_Z), and available water holding capacity of the soil layer (SOL\_AWC) (Table 2). For sediment simulation, the linear parameter of sediment routing capacity (SPCON), CN2 and the Manning's roughness coefficient (CH\_N) are the most sensitive parameters, highlighting the importance of channel processes to sediment yield (Table 2). Unlike flow and sediment, the predicted pesticide load is greatly affected by parameters associated with plant canopy (BLAI, CANMX) and chemical properties (HLIFE\_S, SKOC). In general, the transport and fate of both OP pesticides were determined to a great extent by surface runoff generation and physico-chemical properties.

#### 2.5. Model calibration

The SWAT model was calibrated manually following the sensitivity analysis for Orestimba Creek using data from 2000 to 2006. Monthly model predictions of stream flow, sediment loads and loads of two OP pesticides, chlorpyrifos and diazinon, were compared with the observed data (Tables 3 and 4). Model evaluation statistics for both monthly and yearly simulation indicate good agreement between model prediction and observed values with the Nash-Sutcliffe (NS) coefficient of 0.824 and 0.921 for chlorpyrifos and diazinon, respectively (Table 3). Monthly and yearly flow simulation showed

## Table 2

Ranking of parameters by sensitivities.

a close match to observed values with the NS coefficient of 0.642 and 0.915, respectively (Table 3). The NS values for sediment calibration were above 0.749 and 0.879 for monthly and yearly, respectively (Table 3). SWAT prediction was able to capture the temporal variation in both OP pesticides, with coefficient of determination ( $\mathbb{R}^2$ ) and NS values over 0.82. Mean pesticide loads simulated by the model during the validation period were in good agreement with those measured (Table 4). This suggests that the SWAT model was able to capture the variations in stream flow, sediment load and pesticide load at the watershed outlet.

## 2.6. Baseline simulation

The effectiveness of BMP implementation was defined as the percent change between model outputs predicted from the baseline and from BMP scenarios. The baseline values for input parameters are often selected by either model calibration procedures or a "suggested" value obtained from the literature or user experience (Arabi et al., 2004, 2006). In this study, parameter values were set from the manual calibration processes. The baseline simulation assumed no BMP implementation in the watershed.

Crop management practices including planting, harvesting, fertilization and irrigation were set according to common practices that were identified through consultation with local growers in the Orestimba Creek watershed. Alfalfa was cut seven times per season, with 2–3 irrigation events between cuttings. Almonds were irrigated every 15 days from April through mid-August. Beans and other vegetated crops were assumed to be irrigated every 6–8 days during growing season. Nutrients were applied on the field using the autofertilization module of SWAT model, which monitored plant stress by nutrients and applied fertilizers when the plant was stressed (Neitsch et al., 2005). Pesticide use information was obtained from the PUR database, which provides a very close estimate of the pesticide application timing and amount during 2000 and 2006.

## 2.7. Representation of BMPs

Based on previous literature review and the local agricultural context of Orestimba Creek watershed, four BMPs were selected for testing in this study: sediment ponds, vegetated ditches, buffer strips, and pesticide use reduction. They were selected based on their ease of

Parameter name	Stream flow	Sediment	Chlorpyrifos dissolved	Chlorpyrifos adsorbed	Diazinon dissolved	Diazinon adsorbed
SCS curve number (CN2)	1	2	1	1	1	1
Soil evaporation compensation factor (ESCO)	2	4	6	5	8	7
Fraction of ground water recharge to deep aquifer (RCHR_DP)	3	7	12	13	7	8
Soil depth (SOL_Z)	4	10	10	11	6	6
Available water holding capacity (SOL_AWC)	5	12	14	14	10	10
Average slope steepness (SLOPE)	6	13	16	16	17	17
Saturated hydraulic conductivity (SOL_K)	7	15	15	15	15	15
Maximum canopy storage (CANMX)	8	16	4	9	9	9
Maximum potential leaf area (BLAI)	9	14	2	3	11	12
Threshold depth of water in the shallow aquifer required for return flow to (GWQMN)	10	18	20	20	13	14
Surface runoff lag coefficient (SURLAG)	12	5	13	10	3	3
Effective hydraulic conductivity in main channel alluvium (CH_K2)		6	5	4	2	2
Biological mixing efficiency (BIOMIX)		17	7	8	12	11
Manning's "n" value (CH_N)		3	11	12	16	16
Linear parameter for calculating the maximum amount of sediment that can be		1	23	23	24	24
reentrained during channel sediment routing (SPCON)						
Channel cover factor (CH_COV)	20	7	23	23	24	24
Exponent parameter for calculating sediment reentrained in channel sediment routing (SPEXP)	20	9	23	23	24	24
USLE equation support practice factor (USLE_P)	20	11	8	7	5	5
Degradation half-life of the chemical in the soil (HLIFE_S)		25	3	2	4	4
Soil adsorption coefficient normalized for soil organic carbon content (SKOC)		25	9	6	14	13

#### Table 3

Calibration of the SWAT model for simulation of stream flow, sediment load, chlorpyrifos load and diazinon load.

		Calibration		Validation
		Monthly	Yearly	Monthly
Flow	R <sup>2</sup>	0.747	0.978	0.821
	NS	0.642	0.915	0.809
	RMSE	0.509	1.366	0.504
Sediment	R <sup>2</sup>	0.763	0.963	0.888
	NS	0.749	0.879	0.881
	RMSE	1021	2866	516
Chlorpyrifos	R <sup>2</sup>	0.873	0.963	-
	NS	0.824	0.917	
	RMSE	23.5	43.1	
Diazinon	R <sup>2</sup>	0.995	0.989	
	NS	0.921	0.886	
	RMSE	15.3	64.7	

R<sup>2</sup>: Coefficient of determination

NS: Nash-Sutcliffe coefficient.

RMSE: Root mean squared error.

adoption by growers, their potential effectiveness, and the viability of associated processes in the SWAT model.

#### 2.7.1. Sediment ponds

Previous research indicates that sediment ponds can play an effective role in reducing sediment load and pesticide runoff from agricultural fields (California Stormwater Quality Association, 2003). However, it is unknown whether sediment ponds can also reduce the runoff of OP pesticides, particularly chlorpyrifos and diazinon. The SWAT model provides options for simulating on farm water ponds. The model calculates transport and fate processes for sediments and nutrients within a pond, but not for pesticides. Fortunately, the model does provide a pesticide transport and transformation module for lakes and reservoirs. Therefore, the algorithms for pesticide partitioning and transformation from the lake and reservoir modules were used to simulate pesticide processes in sediment ponds. Similar to the pesticide channel process, the algorithm first partitions pesticides into soluble and adsorbed phases. Pesticides in both adsorbed and dissolved phases were subject to degradation and pesticides in the dissolved phase were subject to volatilization (Neitsch et al., 2005).

A hypothetical sediment pond was designed according to the National Conservation Practice Standard by USDA NRCS (USDA, 2007). Pond sizes were calculated according to the standard with an operation depth of 2.44 m. Areas of the pond ranged from 1400 to 2750 m<sup>2</sup> for holding times varying from 12 to 30 h and a source area of 1000 ha.

## 2.7.2. Vegetated ditches

Vegetated ditches reduce pollutants by increasing the channel roughness, sedimentation and pollutant adsorption to plant surfaces. Therefore, the parameters of channel roughness coefficient, channel erodibility and channel cover were increased to represent this BMP. In this study, we varied the channel roughness coefficient from 0.001 to 0.5 to reflect a full range of vegetation cover conditions. The value

#### Table 4

Comparison of observed and predicted values for chlorpyrifos and diazinon load during model validation. (Unit: g).

		Observed	Predicted
Chlorpyrifos	Mean	43.1	44.6
	STD	30.1	77.5
Diazinon	Mean	24.4	33.3
	STD	23.3	4.6

<sup>a</sup>Only mean and standard deviation (STD) were compared due to limited data available for validation.

range was selected according to the USGS guidance in selecting Manning's roughness coefficient (Arcement and Schneider, 1984).

#### 2.7.3. Buffer strips

Vegetated buffers are designed to use vegetation to remove sediment, nutrients and pesticides from surface water runoff through filtration, deposition, adsorption and infiltration (Dillaha, 1989). The SWAT model uses a conservative filter strip trapping efficiency to calculate the mass of sediment, nutrients and pesticides that is trapped by the filter strip. The trapping efficiency is calculated as:

$$trap_{ef} = 0.367 \times \left( width_{buffer} \right)^{0.2967} \tag{2}$$

where  $trap_{ef}$  is the fraction of pollutant mass trapped by the filter strip, and  $width_{buffer}$  is the width of the filter strip (m). In this study, buffer widths of 5, 10, 15, 20, 25, 30, 35, 50, and 100 m were tested to evaluate the effects of buffer width.

## 2.7.4. Pesticide use reduction

The most effective and straightforward way to reduce pesticide pollution is to minimize pesticide application. Various approaches can be taken to reduce the use of OP pesticides. For instance, integrated pest management practices provide various alternative management practices for OP use (Zhang et al., 2008). Examples of the alternative management practices include pest pressure monitoring to avoid dormant-season application, biological control, and the use of reduced risk pesticides. In addition, smart sprayer technologies can be used to increase spray precision and reduce total pesticide use. This study tested the scenarios of pesticide use reductions ranging from 5 to 50% of the current use amount for all the crops.

#### 2.7.5. Combined BMP scenarios

BMPs may achieve higher efficiency when used in combination (Osmond et al., 1995). To examine the effects of combined BMPs, five combinations of BMPs were studied. Individual BMPs used in the combinations was "designed" to achieve reasonable balance between effectiveness and cost. Use reduction was set at 15%, water holding time for sediment ponds was set at 24 h; width of buffer strips was set at 20 m and the Manning's roughness coefficients for vegetated ditch was set to 0.15. As a result, there were a total of 10 BMP implementation scenarios simulated in this study (Table 5).

## 3. Results and discussion

#### 3.1. BMP effectiveness

#### 3.1.1. Sediment pond

Simulated results showed that sediment ponds were effective in removing sediment from agricultural runoff. Sediment load was reduced by about 58% compared to the baseline scenario (Fig. 2). This number was comparable to the findings in previous studies (Fiener et al., 2005; Markle, 2009; McCaleb and McLaughlin, 2008). An 8-year field monitoring study by Fiener et al. (2005) found that detention ponds trapped 54-85% of sediment from field runoff. Markle (2009) demonstrated the effectiveness of a sediment pond in a California almond orchard. Their experiments showed 80-84% removal of sediment by the pond. The sediment ponds used in those field studies were generally much smaller than our hypothetical pond that was "designed" to treat a much larger source area. This partially explains why the simulated efficiency was relatively low compared to the field experiments. In contrast, sediment ponds did not show a significant impact on surface flow. Predicted stream flow was reduced slightly (0-0.3%) compared to the baseline simulation (Fig. 2). This is not surprising because the pond bottom hydraulic conductivity was only 0.001 mm/h by default, which resulted in a low infiltration rate.

## Table 5

BMP scenarios simulated in the study.

BMP scenarios	Abbreviations
Sediment pond	SP <sup>a</sup>
Vegetated ditch	VD <sup>b</sup>
Buffer strip	BS <sup>c</sup>
Use reduction	UR <sup>d</sup>
Vegetated ditch + Sediment pond	VD + SP
Vegetated ditch + Buffer strip	VD + BS
Use reduction + Buffer strip	UR + BS
Use reduction + Sediment pond	UR + SP
Use reduction + Vegetated ditch	UR + VD
Use reduction + Vegetated ditch + Buffer strip	UR + VD + BS
Use reduction + Vegetated ditch + Sediment pond	UR + VD + SP

<sup>a</sup> Holding time = 24 h, depth = 2.44 m, and pond surface area = 2340 ha.

<sup>b</sup> Manning's n = 0.15.

<sup>c</sup> Buffer width = 20 m.

<sup>d</sup> 15% reduction from current use.

Overall, sediment ponds were more effective in removing chlorpyrifos than diazinon. A 12-hr treatment removed 78% of total chlorpyrifos as opposed to 28% of total diazinon. This difference is likely due to different sorption properties of these two chemicals. Chlorpyrifos is more readily attached to sediment compared to diazinon, resulting in a higher removal efficiency of this chemical by sediment ponds. This suggests that sediment ponds are more effective in removing hydrophobic pesticides than those with low soil-water partition coefficients (Koc). The finding was further signified by the differences of the ponds' effectiveness in removing adsorbed and dissolved pesticides (Fig. 2). The pond removed about 27–44% of the adsorbed pesticides, 3–50 times higher than those of the dissolved forms (2–10%).

Previous studies have shown that the trapping efficiency of sediment ponds was associated with retention time (Brown et al., 1981, Edwards et al., 1999). Yet, the effects seem to vary among different constituents (Fig. 2). Extending retention time beyond 12 h posed little impacts on trapping efficiency for sediment, but significantly changed those for adsorbed and dissolved pesticides. The pond's sediment trapping efficiency remained steady after 12 h, mainly because the majority of the sediment was retained within the first 12 h (Edwards et al., 1999). Within a sediment pond, large coarse particles easily settle to the bottom while fine sediment tends to remain in the water column (Budd et al., 2009, Fiener et al., 2005). Once large particles settle out of the water column, which normally occurs within 12 h, increasing holding time may not further increase sediment removal.

While prolonged retention time decreased trapping efficiency for adsorbed chemicals, it increased the efficiency for dissolved chemicals. This is likely due to the dynamics of sorption and re-suspension processes within the pond. With increased residence time, adsorbed



Fig. 2. Effectiveness of sediment ponds with varying holding time. Positive efficiency values indicate less pollutant mass in the output of BMP scenarios than that of the baseline simulation.

pesticides may be released from sediment to the water column via resuspension and desorption, decreasing the overall trapping efficiency. On the other hand, longer retention time allows dissolved pesticides to attach to large particles and later to settle to the bottom, increasing the overall trapping efficiency. Since the simulated retention times were relatively short compared to the pesticides' half-lives, degradation effects were considered negligible. Yet in reality, extended residence time may show a positive impact on pesticide removal due to pesticide degradation and transformation. Numerous field experiments are needed to identify the mechanisms of pesticide removal by sediment ponds.

Although sediment ponds were considered a promising mitigation tool for reducing ANPSP, they may not play an important role for OP pesticide removal. This simulation study showed that sediment ponds can only reduce dissolved diazinon and chlorpyrifos by less than 10%. This finding was consistent with previous studies. Moore et al. (2007) studied a constructed wetland system comprising a sediment pond cell and two vegetated wetland cells. Their results on mass distribution of diazinon indicated that the sediment retention pond played a minor role in overall wetland retention of diazinon. Another study conducted by the California Department of Pesticide Regulation also confirmed that sediment ponds did not reduce diazinon concentrations in water, although they did slightly for chlorpyrifos (Walters et al., 2000). However, the fact that sediment ponds removed about 58% of sediment suggested that they may be effective in reducing sediment bound hydrophobic pesticides such as pyrethroid pesticides, which have lower solubility and greater tendency to adsorb to organic carbon in sediment (Budd et al., 2009). A recent field study by Budd et al. (2009) indicated that sediment ponds reduced the total pyrethroid load by 52-94%. In addition, while extended retention time posed mixed impacts on pesticide removal depending on their chemical speciation (dissolved or adsorbed), it may eventually reduce pesticide load through degradation and transformation. Despite their ineffectiveness in removing dissolved pesticides, sediment ponds may still play an important role when used in combination with other BMPs. Since they settle out most of the sediment, these ponds may prevent sediment from building up in the downstream flow paths of other BMPs, such as vegetative ditches.

#### 3.1.2. Vegetated ditches

Vegetated ditches function by increasing channel roughness with vegetation cover. Fig. 3 shows the impacts of channel roughness on various constituents. Increasing channel roughness reduced the masses of diazinon and chlorpyrifos in both dissolved and adsorbed phases. A vegetated ditch with Manning's roughness coefficient of 0.075, which is the median roughness coefficient for channels with dredged ditches covered by un-maintained weeds and brush (Chow, 1959; Neitsch et al., 2005), reduced over 20% of sediment, diazinon and chlorpyrifos loads. Increasing channel roughness slows down runoff flow and consequently lengthens residence time.

Many physical and chemical processes contribute to the removal of pesticides by a vegetated ditch. The main processes include sedimentation, infiltration and adsorption to plant surfaces. The impacts of increased channel roughness on sedimentation and infiltration were simulated in the SWAT model. Pesticide adsorption to plant surfaces was not reflected in the model because this mechanism was not well characterized in the literature. The relative importance of these processes was unknown, but studies have shown that plants play an important role in trapping sediment and pesticides within a vegetated ditch system (Moore et al., 2008; Rogers and Stringfellow, 2009). Laboratory experiments conducted by Rogers and Stringfellow (2009) suggested that chlorpyrifos sorption to plant stems was more than 10 times higher than to soil. Simulation of pesticide sorption to plants within a vegetated ditch may become possible in the future when the sorption coefficients are better quantified.



Fig. 3. Effects of channel roughness on pesticide load.

In Orestimba Creek watershed, orchards and farms are usually equipped with dredged ditches for conveying irrigation return flow. To ensure that the ditches deliver tailwater at a timely manner, growers often apply herbicides to keep the ditch clear. However, this study and many others in the literature have shown that creating a vegetation cover could trap sediment and pesticides in tailwater thereby preventing pesticides from entering the surface water system (Moore et al., 2000; Cooper et al., 2004; Bouldin et al., 2005).

## 3.1.3. Buffer strips

Buffer strips are effective in reducing the simulated constituents (Fig. 4). A five-meter buffer reduced sediment and OP pesticides by 37% and 59%, respectively. Widening the buffer to 25 m increased the removal effectiveness to 56% and 89% for sediment and OP pesticides, respectively. These results are consistent with many field studies on filter strips. Vianello et al. (2005) studied the effects of vegetative filter strips in Po Valley, Northeast Italy, and found that herbicide runoff could be reduced by 86–98% during runoff events. Watanabe and Grismer (2001) studied the effects of vegetative buffers on attenuating diazinon in a California orchard and found that the diazinon load in treatments with vegetative filter strips was only about one-fourth as much as in untreated control fields.

Vegetated buffers have been considered as an important mitigation tool for ANPSP. Simulating their effectiveness is essential to the design and implementation of buffers for successful mitigation. Current research direction points to linking empirical equations with hydrological models (Fox and Sabbagh, 2009). SWAT takes such an approach. Pollutant runoff from adjacent fields was routed through the vegetated buffer and reduced by the amount calculated from an empirical equation (Eq. (2)). The empirical equation used in SWAT calculates the buffer's trapping efficiency as a function of buffer width. This practice provides reasonable predictions on buffer effectiveness; however, there is much room for improvement. A meta-analysis study by Zhang et al. (2009) suggested that an exponential equation in the form of  $Y = K \cdot (1 - e^{-b \times bufferwidth})$ ,  $(0 < K \le 100)$ , may be a better quantification than Eq. (2). Eq. (2) is an empirical equation based on analysis performed using field studies, which mainly focused on nutrients and sediment (Neitsch et al., 2005). However, the model developed in Zhang et al. (2009) was derived from a theoretical base and tested against experimental data for pesticides. Furthermore, recent studies have revealed that although being the dominant factor, buffer width was not the only variable determining a buffer's trapping efficiency (Reichenberger et al., 2007). Models developed with additional variables such as slope, vegetation type, and pesticide physicochemical properties may provide improved results. In addition, processes based models such as the VFSMOD-W (Muñoz-Carpena et al., 2010) model, which considers incoming runoff depth and infiltration may be another viable alternative.

### 3.1.4. Use reduction

The effects of reducing pesticide application rates on pesticide instream load were close to linear (Fig. 5). A 15% reduction of the current pesticide use would result in a load reduction of at least 28% and 26% for diazinon and chlorpyrifos, respectively (Fig. 5). The change rate for chlorpyrifos is more pronounced than diazinon mainly because the use amount of chlorpyrifos is higher. Guo et al. (2004) suggested that pesticide use amounts and rainfall are the most important factors determining pesticide load in waterways. Pesticide use reduction is one of the most important approaches for reducing pesticide load in surface waters. In addition, integrated pest management practices should be used on orchards and fields to minimize pesticide use whenever possible. In cases of high pest pressure, alternative lower risk pesticides may be used to replace the OP pesticides. When OP application is necessary, smart sprayers may be used to increase spray efficiency.

Previous studies indicated that the application timing was another important factor in determining pesticide load (Chu and Marino, 2004; Luo et al., 2008). Use during the rainy season had greater impacts on pesticide runoff compared to use during the dry season (Luo et al., 2008). This study assumes a uniform use reduction throughout the year. Therefore, reducing pesticide use during the raining season may yield greater reductions in pesticide loads than those predicted by the model.

#### 3.1.5. Combined BMP scenarios

In addition to the four above mentioned BMPs, this study investigated the effectiveness of combining BMP scenarios. Since sediment ponds and buffer strips both occupy additional farm land, it is highly unlikely that growers would implement these simultaneously on the same site. Therefore, the combined use of sediment ponds and buffer strips was not analyzed. As a result, in addition to the four individual BMPs, seven combination scenarios were simulated and compared with the baseline simulation. Fig. 6 show the effectiveness of the 11 individual and combined BMP scenarios. The combination of use reduction with



Fig. 4. Effects of buffer width on pollutant mass.



Fig. 5. Effects of pesticide use reduction on pesticide load.



Fig. 6. Effectiveness of BMP scenarios in removing simulated constituents. Abbreviations: SP: Sediment pond; UR: Use reduction; VD: Vegetated ditches; and BS: Buffer strip.

vegetated ditches and buffer strips (UR + VD + BS) showed the highest efficiency in removing dissolved diazinon and chlorpyrifos, followed by buffer strips with vegetated ditches (VD + BS) and buffer strips with use reduction (UR + BS). The scenario of UR + VD + BS removed over 60% and 94% of sediment and OP pesticides, respectively (Table 6). A 20 m buffer strip alone removed over 55% of sediment and 89% of both OPs (Table 6). Effectiveness of individual BMPs followed the rank of buffer strips (BS)> vegetated ditches (VD)> use reduction (UR)> sediment ponds (SP). It should be noted that the rankings were only as true as the assumptions and settings of each BMP. For example, a use reduction higher than the assumed 15% may result in greater removal effectiveness than a 20 m buffer. In general, sediment ponds seemed to be the least effective for removing OP among all the BMPs, but it was the most effective for removing sediment.

Mitigating ANPSP in a watershed requires combinations of various BMPs. The simulated results of combined BMPs revealed the promise of this approach. The BMPs included in this investigation target different stages of pesticide runoff: use reduction controls the source of the pollution, sediment ponds and buffer strips function at the edge of the field, while vegetated ditches trap pollutants during transport off site. Combining these BMPs produced additive effects in trapping sediment and pesticides from surface runoff.

## 3.2. Future modeling improvements

In general, the SWAT model produced reasonable predictions of BMP effectiveness. However, limitations still exist in simulating BMP effectiveness using the SWAT model. First, even though the

 Table 6

 Percent reductions of flow and pesticide loads by different BMP scenarios.

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	Scenario code	Flow (%)	Sediment load (%)	Dissolved diazinon (%)	Adsorbed diazinon (%)	Dissolved chlorpyrifos (%)	Adsorbed chlorpyrifos (%)
	UR + VD + BS	0.0	60.5	94.8	94.5	95.5	94.4
	VD + BS	0.0	60.5	92.7	92.3	93.8	92.2
	UR + BS	0.0	55.8	92.2	92.4	92.0	92.9
	BS	0.0	55.8	89.1	89.2	89.2	89.2
	UR + VD	0.0	30.3	52.2	49.8	59.0	47.8
	UR + VD + SP	0.0	62.3	50.3	72.7	53.6	71.4
	VD	0.0	31.4	33.4	30.0	44.3	29.0
	VD + SP	0.0	62.3	30.5	61.8	36.2	60.6
	UR	0.0	0.0	28.1	29.4	26.4	34.1
	UR + SP	0.1	59.7	22.4	59.9	16.0	63.6
	SP	0.3	58.6	4.9	31.1	3.0	40.9

Abbreviations: UR: use reduction; VD: vegetated ditches; BS: buffer strip; and SP: sediment pond.

SWAT model includes a large number of parameters, parameters representing BMP implementation are limited. For example, the module for calculating buffer strip efficiency includes only buffer width as an adjustable parameter. Recent studies indicated that besides buffer width, slope and vegetation type play an important role (Reichenberger et al., 2007; Zhang et al., 2009). More realistic representations of BMPs can be achieved by including additional parameters reflecting the changes in pesticide movement as a result of BMP implementation. Second, SWAT model's abilities to simulate various management practices and their impacts on the watershed make it one of the best available models for simulating BMP effectiveness. But it does not simulate processes at the field scale, the scale at which most of the BMPs are physically implemented. The lack of spatial definition within the HRU areas creates another limit for the model's ability to simulate BMPs. For example, the model simulates one sediment pond for each subbasin in the sediment pond simulation. Water, sediment and pesticides were collected from the land phase, routed through the pond and then routed to the stream network. While many BMPs, such as buffer strips, function at the field edge, the model generates output at the watershed outlet. Therefore, the model outputs not only include results from BMP implementation but also results from processes occurring between the field edge and the watershed outlet. This introduces additional uncertainties to the model output. A possible solution to the problem is suggested to link SWAT with a field-scale model. A current effort is the APEX-SWAT model, which links the field-scale based APEX (Williams et al., 2000) model with the SWAT model. The APEX-SWAT model accumulates the field outputs of the APEX model for a subbasin and routes them through the channel network in the SWAT model (Saleh and Gallego, 2007). However, the model needs to be tested for its capabilities of effectively representing BMPs. Finally, the current practice assumes a fixed value for the representative parameters for each BMP. Potentially, a probabilistic approach using Monte Carlo simulation can be applied to derive the probability density function (PDF) of BMP effectiveness using the distributions of input parameters. This approach has been previously applied to derive the PDFs of pesticide concentrations based on monitoring data (Spurlock et al., 2005, 2006). This approach requires a better understanding of BMP mechanisms so that distributions of their representing parameters can be obtained. As more field experiments on BMP effectiveness become available, this may become possible in the future.

## 4. Conclusions

Overall, the SWAT model provided a viable platform to simulate BMP effectiveness at the watershed scale. Model predictions on effectiveness of individual BMPs and their combinations were consistent with previously reported field studies. Simulation results suggested that combining use reduction with vegetated ditches and buffer strips was the most effective method for removing sediment and OP pesticides. Buffer strips and vegetated ditches were also effective in reducing OPs. Sediment ponds, however, were only effective in reducing sediment and adsorbed OPs, but not dissolved OPs. The results will assist decision making in implementing BMPs to reduce pesticide loads in surface runoff.

This study not only provided a framework of modeling the effectiveness of both individual and combined BMP scenarios, but also identified data gaps for future investigations. Future research should be directed toward the improvement of model algorithms to facilitate BMP simulation, i.e. including more adjustable parameters to represent BMP mitigation processes and improving the equations for calculating buffer strip efficacy with a better model. In addition, uses of Monte Carlo simulation will also improve the determination of BMP effectiveness by sampling parameter inputs from their distribution, as opposed to assigning a single value.

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