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APEX simulation: Water quality of Sacramento Valley wetlands impacted by waterfowl droppings

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Abstract: Since most wetlands in the Sacramento Valley of California are dependent on artificial water delivery, supplying water for wetland management is the greatest challenge to wetland managers, especially during drought years. Efforts are needed to improve the security of water supplies for optimal habitat management and water quality improvement. This study contributes to these efforts by developing an eco-hydrologic model (Agricultural Policy/Environmental Extender [APEX]) of this wetland system, which has key components evaluated in the wetland simulation, including wetting and drying of wetland soils, competition and response of wetland species to wetland hydrology, settling of sediment, and nitrogen (N) removal. APEX model calibration (April of 2017 to May of 2018) and validation (June of 2018 to August of 2018) resulted in a percentage bias (PB) of 9.8% and –8.5%, respectively, for total volume of water holding in four serially connected wetlands. The N contents in the wetland waterbody were calibrated and validated using the monitored values collected during 2017 to 2018 and 2015 to 2016, respectively. All PB values for calibration and validation were over 35%. The calibrated model was used to evaluate the effects of wetland management and increasing temperature on N removal. Moreover, an additive regression model (ARM) was developed based on bird survey data and used to analyze bird dropping seasonal patterns and access their impacts on water quality in the studied wetlands. Based on the results of the model, the wetland water quality was influenced by waterfowl populations and eventually governed by water availability in each wetland cell. The N removal by wetlands was negatively affected by the volume of irrigation water. Moreover, increasing temperature caused a decrease in waterfowl population, which led to decreased N concentration by up to 42%. Overall, the results indicate that the developed model can be effectively used to quantify the effects of wetland management on water balance, water quality, and vegetation and to describe the nexus of wetland management, water use, and ecosystem service functions of managed wetlands.

Key words: additive regression model—climate change—irrigation—nitrogen—wetland

Wetlands are critical features in the landscape that provide a range of valuable ecosystem services to benefit agriculture and human wellbeing. They help reduce the impacts from storm damage and flooding, maintain good water quality, increase groundwater recharge, store carbon (C), act as a nutrient sink to improve water quality downstream, and provide food and habitat for fish and wildlife (Smith et al. 2015). However, wetlands have been steadily and rapidly disappearing since 1900 with more than 60% of wetlands in the United States destroyed or degraded (Wright et al. 2006). The irreversible loss of wetlands resulted from land-use changes due to extensive development in urban and rural areas (Bedford et al. 1999; Dahl 2014; Ramsar 2015). For example, 60% to 65% of small wetlands such as prairie potholes have been drained to facilitate crop production (Dahl 2014). Most wetlands surrounded by urban land use have been heavily degraded due to changes in water quality, quantity, and flow rates. Increases in pollutant inputs are causing the death of plants and aquatic animals (USEPA 2008). To maintain the benefits of wetlands and their surrounding ecosystems, the importance of wetland conservation and restoration has been strongly recognized (Kentula 2002). The current goal of the US government is to prevent a net loss of wetlands through conserving existing wetlands and restoring lost wetlands (SFSA 2015).

Wetland monitoring and assessment programs use several tools for states and tribes to better manage and protect wetland resources (USEPA 2006). Wetland monitoring and assessment programs are designed to analyze monitoring data as well as to evaluate conditions and functions of wetlands. These support regulatory decision-making and planning (USEPA 2006). Wetlands are characterized by three factors: hydrology, soils, and vegetation (Fennessy et al. 2007). These three variables are causally related, so information on one factor can be used to support another (CCW 1995). All three variables are used as indicators in numerous effective wetlands monitoring and assessment strategies including quantitative biological assessments (USEPA 2003) and hydrogeomorphic (HGM) functional assessments (Cole 2017). These have been used to assess wetlands at a variety of spatial scales (Collins et al. 2007). Biological assessments are used to evaluate wetland health by direct measurements of the condition of biological attributes such as taxonomic richness, community structure, and health of individual organisms (USEPA 2002). Biological assessments are used for validating or calibrating rapid assessment methods that are most commonly used by state and regional wetland

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managers because they require a relatively small investment of time and effort (Fennessy et al. 2007). Once validated, rapid assessment methods provide quantitative information on the condition and function of wetlands. The HGM approach is also developed to facilitate longer term development of rapid assessment methods of estimating wetland functions (Brinson 1996). Also, the HGM approach is used for wetland classification (Brinson 1996; Brooks et al. 2011) that can assess wetland hydrologic function without actually collecting field data (Cole 2017).

Wetlands of California's Sacramento Valley are distributed in the northern Central Valley and play an important role in maintaining habitats for a variety of diverse wildlife, particularly migratory birds of the Pacific Flyway. While more than 230 species, including many birds, use the Sacramento Valley annually, others occur there only during migration, for the winter, or during the spring and summer months to breed (US FWS 2009). Ninety to ninety-five percent of freshwater wetlands in California's Central Valley have been lost or degraded due to agricultural and urban development (Frayer et al. 1989). Over recent decades, the remaining wetlands have been often intensively managed by federal and state agencies, nongovernmental organizations, and private individuals to maximize resources to support abundant migratory birds, endangered and threatened species, and other wetland-dependent wildlife (US FWS 2009). Most managed wetlands are maintained through artificial flooding and draining during specific time periods utilizing dikes, water control structures, and pumps (US FWS 2009). However, persistent dry conditions over the past few years have put the Sacramento Valley's wetlands under severe water stress (Robenson 2015).

Water scarcity, coupled with increasing demands for water by cities and farmlands, has led to decreased water availability for wetlands and the subsequent loss of plants and wildlife habitat. To meet water demand in wetlands, agricultural wastewater (e.g., irrigation tailwater and subsurface drainage) has been discharged into the existing, natural wetlands (Lemly 1994). However, wastewater often has many quality concerns, such as pathogens, heavy metals, elevated salinity (or sodicity), ammonia, pesticides, or pharmaceuticals. This raises short- and long-term soil and human/animal health concerns. Inorganic nutrients (e.g., nitrogen [N] and phosphorus [P]) from livestock manure also increase the risk of contaminated wetlands (Berg et al. 2017). According to a water quality survey from 17 wetlands located across the Central Valley (Kahara and Duffy 2016), after waterfowl migration, total N loads in water outflow from wetlands significantly increased compared to water inflow. Identification of the quantity and quality of water required to sustain ecosystem health, therefore, is necessary for prioritizing conservation actions and providing guidance to agencies responsible for balancing human needs and ecosystem requirements. Soil and water modeling system tools quantify values of functions and process of wetland ecosystems, such as nutrient cycling, species richness, productivity, water quality, and rates of organic accumulation.

The Agricultural Policy/Environmental Extender (APEX) model, a field-scaled process-based biophysical model, has emerged as a valuable tool that effectively evaluates hydrologic and water quality functions of small closed-basin depressional wetlands (Mushet and Scherff 2017). The APEX model is a multifield version of the processor Environmental Policy Impact Climate (EPIC) model. APEX has additional algorithms to simulate water quality, N and C cycling, and plant growth. APEX simulates the impacts of different wetland conservation programs and practices on surface runoff and losses of sediment and nutrients across complex landscapes and channel systems to the watershed outlet (Wang et al. 2012). Also, APEX simulates flow and pollutant transport at the watershed scale. In the present study, a wetland functional assessment model using APEX was developed to simulate and assess water quantity and quality effects of artificially managed wetlands. These wetlands provide up to half of its annual water supplies for environmental flow by extending recent developments of APEX (Choi et al. 2017; Sharifi et al. 2019), which will help to improve Pacific Flyway habitat for migratory birds and other native species (Kahara and Duffy 2016). The developed model system helps our understanding of the potential impacts of the artificial wetlands on peak stream flows, net transport of sediment and pollutants after water irrigation, net nutrient accumulation from waterfowl droppings, and the role of plant density and plant community with respect to nutrient uptake.

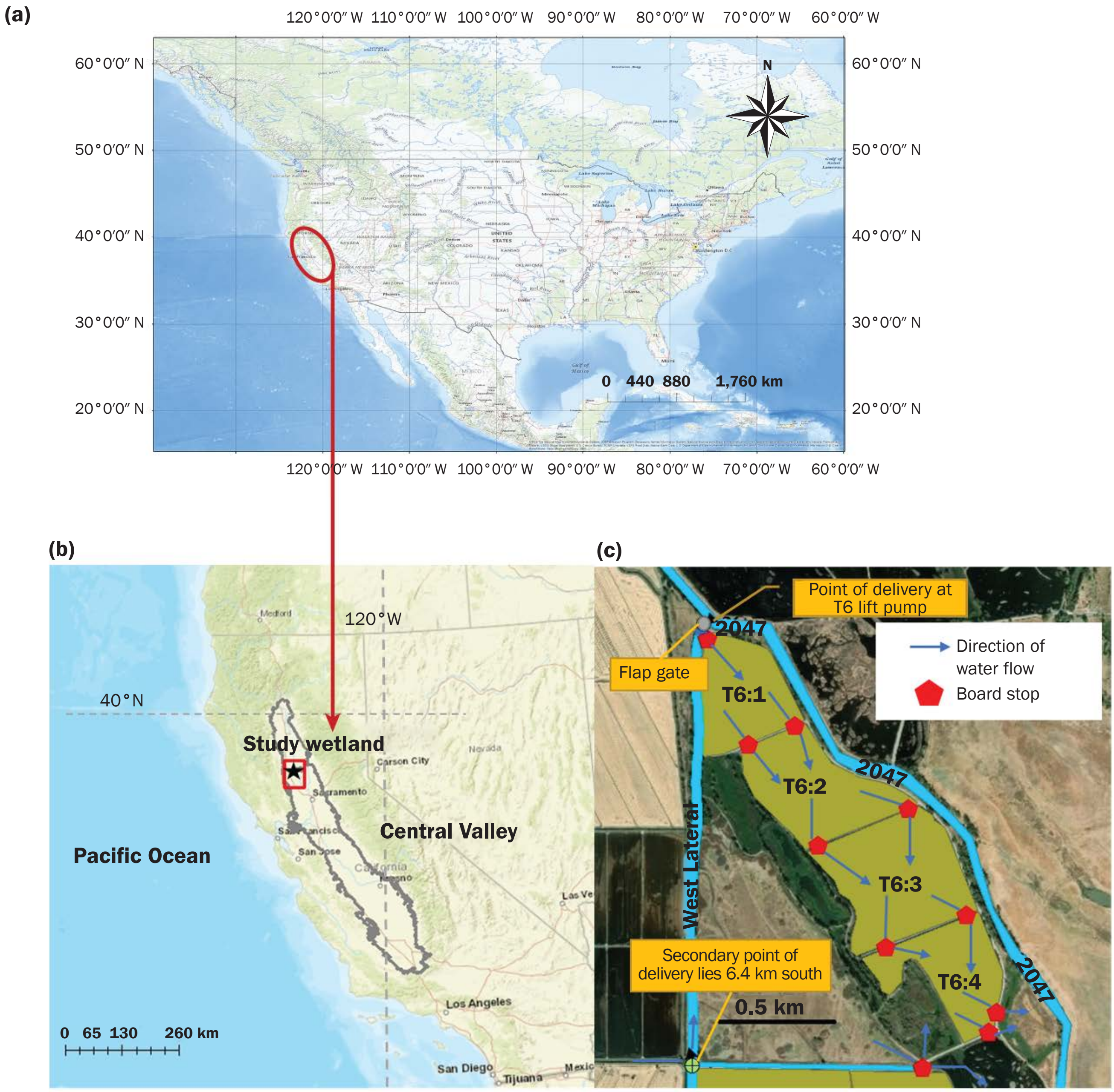
This study focused on research for facilitating sustainable wetland management in California's Sacramento Valley. For this purpose, a hydrological model has been developed to assess various biophysical processes occurring in wetlands. APEX models were constructed and calibrated for an artificially managed wetland located in California's Sacramento Valley for which water flow and water quality were monitored for four years (2015 to 2018). The results will provide useful information that addresses water quality problems on a watershed level and technical suggestions to effectively reduce certain types of nonpoint source (NPS) pollution for planners and decision-makers.

Materials and Methods

Study Area. The APEX model was applied to an artificially managed wetland at Colusa National Wildlife Refuge (CNWR), which is part of California's Sacramento Valley (figures 1a and 1b). Kahara and Duffy (2016) provided a detailed description of the wetland. The study wetland, Tract 6 (T6), is an irrigated seasonal wetland that is supplied a brief summer irrigation and is later inundated from fall to spring. T6 is divided into four cells of ponding areas (T6:1, T6:2, T6:3, and T6:4), which are hydrologically connected but differ in size. The areas of T6:1, 2, 3, and 4 are 10.9, 13.8, 18.2, and 11.7 ha, respectively. All cells are connected through weir-culvert pairs construed in the berms (figure 1b). The soil type in the wetland is Willows Silty Clay soil primarily composed of 53% clay and 44% silt. The T6 wetland has mixed community of marsh vegetation and grain crops. According to a vegetation coverage survey conducted by US Fish and Wildlife Service (2016), watergrass (*Echinochloa crusgalli*) and annual smartweed (*Polygonum hydropiper*) are dominant species in all four cells.

Much of the surrounding lands are farmland. Some wetlands in the area receive runoff from agricultural fields. The West Lateral Canal (WLC) is in Glenn-Colusa Irrigation District, and the Sacramento River serves as the principal water source for the district. Water from the WLC flows north from the southwest corner of CNWR and then flows through the units and other canals to the east and south. The WLC delivers relatively clean water, which is pumped from the Sacramento River, north of Hamilton City, California. The main agricultural drainage canal for the west side of the Sacramento Valley is the Colusa Basin Drain/No. 2047 Canal. The agricultural drainage water from

Figure 1
(a) Location of Central Valley in United States, (b) study wetlands (US FWS 2009), and (c) water flow map in study area at Colusa National Wildlife Refuge (CNWR) in California, United States.



the 2047 Canal tends to pick up considerable amounts of sediments and associated nutrients (e.g., P), as well as water-applied agricultural chemicals such as pesticides and N fertilizers (Tanji and Kielen 2002). T6:1 receives water from both WLC and the 2047 Canal, which is passed in sequence to T6:2, T6:3, and finally T6:4 (figure 1c).

Nutrients Added to Wetlands. Nutrient loading by waterfowl to the artificially managed wetland, T6, was estimated from four data sources: (1) the counted number of waterfowls per month; (2) the rate of production of droppings estimated from the literature; (3) total nutrient content of droppings estimated from the literature; and (4) percentage of flooded area within each subwetland. The number of individuals of each bird species and the percentage of the pond's surface area that was flooded was recorded monthly on each cell between 2015 and 2018 by Sacramento NWR Complex biologists as part of the wildlife monitoring protocol (US FWS 2016). They estimated the number of each species of waterfowl, shorebird, and waterbird with 90% confidence interval using standardized survey routes that are designed to provide optimal viewing of each wetland management unit (US FWS 2016). Through sorting bird species by their total dropping mass, nine species were found to be major contributors of droppings (98% of total droppings), and they were considered individually in nutrient loading calculations. The nine species are white fronted goose (*Anser albifrons*), northern pintail (*Anas acuta*), white geese (snow [*Chen caerulescens*] and Ross's geese [*Chen Rossii*] combined), green-winged teal (*Anas carolinensis*), northern shoveler (*Anas clypeata*), mallard (*Anas platyrhynchos*), American wigeon (*Anas americana*), American coot (*Fulica americana*), and gadwall (*Mareca strepera*). The body weights and rates of production of droppings of the major nine bird species were obtained from the previous studies (Palmer 1962; Kear 1963; Sanderson and Anderson 1978; Terres 1987; Scherer et al. 1995). The body weight of each species was multiplied by the number of individuals of each bird counted per month, and the resulting values in kilograms were multiplied by the monthly rates of dropping production of body weight (dry kg kg^{-1}). The N and P concentrations of droppings were assumed to be 2.34% and 1.87% of the dry weight of droppings based on the average of concentration reported for ducks and geese (Paloumpis and Starrett 1960; Kear 1963; Manny et al. 1975; Gould and Fletcher 1978; Sanderson and Anderson 1978; Harris et al. 1981; Bazley and Jefferies 1985; ASAE 1999; Barker and Walls 2002). Total N and P monthly loading in each cell were calculated by multiplying the dropping production with nutrient contents.

Waterfowl migration can heavily be influenced by climate change because it affects wetland health through changes in water level, precipitation pattern, temperature, and plant communities (Browne and Dell 2007). Total N contribution by waterfowl was established using additive regression model (ARM) proposed by Kim et al. (2017) using equation 1:

$$Y = \alpha + f_1(X_1) + f_2(X_2) + f_3(X_3) + \varepsilon, \varepsilon \sim N(0, \textstyle\sum_{j=1}^{3} \sigma_j^2) \ , \tag{1}$$

where

$$f_j(X_j) = \beta_{j1}(X_j) + \beta_{j2}(X_j^2) + \cdots + \beta_{jL}(X_j^L)$$

and

$$\alpha = \textstyle\sum_{j=1}^{3} \beta_{j0} \ .$$

Month of data collection (X_1), maximum temperature (X_2), and total precipitation amount (X_3) were used as predictor variables. Total water amount (m^3) in each cell was calculated by multiplying water depth (m) with the size of area of cell (m^2). Precipitation pattern and plant community factor were removed due to low correlation score to nutrient contents. The total N estimation can be used to accurately estimate nutrient loadings by bird droppings under various climate conditions. Weather data included total precipitation and maximum temperature for the study site from 2015 to 2018, and were collected from the National Oceanic and Atmospheric Administration (NOAA).

The ARM is evaluated via the 10-fold cross validation (CV), which can find the most appropriate polynomial function (i.e., $f_j(X_j)$) explaining impact of a predictor variable on a response variable (i.e., N). In this study, the CV selects a polynomial function with the maximum R^2 value.

Water Data Collection. The water quality data used for model calibration and validation is maintained by the Wildlife Department at Humboldt State University in support of the USDA Natural Resources Conservation Service (NRCS) Conservation Effects Assessment Project (CEAP). During the spring/summer months of May to August from 2015 to 2018, water samples were collected at inlet and outlet of each cell. Water samples were stored at 4°C and analyzed for total nitrogen (TN) at the University of California-Davis Analytical Laboratory. Between April of 2017 and September of 2018, water depth was measured at a daily time-step using a depth transducer and data logger at the inlet and outlet of each cell. In T6:4, there are many missing data because of periodic data logger malfunction. The missing data were not included for model calibration and validation.

In 2015, vegetation coverage and composition were determined using transect method in each wetland. Transects were spaced approximately 50 m apart. The plant density expressed in plants per square meter (plants m^{-2}) was computed simply by averaging the proportional coverage of each species across the number of quadrats surveyed, then multiplying this value by the total unit area. All wetland plant species were classified into three functional groups, including grasses and forbs, based on the following functional traits: maximum leaf area index (Max LAI) and their plant types (Williams et al. 2020). The Max LAI for forbs ranged from 0.54 to 1.88 and were generally lower than grasses' Max LAI, which ranged from 0.98 to 2.40 (Williams et al. 2020).

Model Application. In this study, an enhanced APEX model for simulating surface inundation (Choi et al. 2017) was used to simulate the impact of irrigation management on plant growth, water balance, and water quality in the T6 wetland. The enhanced APEX model simulates ponding in subareas (or land units) by flooding water. The water ponding condition (e.g., water depth) is controlled by an outlet control such as weirs in each subarea. Since the T6 wetland is comprised of four different sized cells, total four APEX subarea modules were created. For this wetland study, the APEX1501 code was modified to allow for draining discharge water from a wetland subarea into the immediate downstream wetland subarea and inundate the pond. To simulate ponding events at the T6 wetland, two different irrigation operations were scheduled each year: between September and April/May (winter/spring) and between April/May and September (summer). The simulated wetland was flooded for only a short time during summers to promote vegetation growth, then managed to hold water from fall to late spring. Due to recurring sequence of irrigation operations between years, we created summer and winter irrigation cycles in the APEX management schedule. The summer and winter irrigation cycles were created based on the collected water depth data from 2017 and 2018 as summarized in table 1. Soil information was downloaded from the USDA NRCS SSURGO database: http://websoilsurvey.nrcs.usda.gov/. Weather from NOAA, https://www.ncdc.noaa.gov/, was used in the simulation.

In summer of 2016 to 2018, vegetation growth parameters such as aboveground biomass, LAI, and light intercept were measured at the T6 wetland. Based on the collected plant data, growth cycle and plant parameters of each wetland vegetation type, including grasses and forbs, were developed through Agricultural Land Management Alternative with Numerical Assessment Criteria (ALMANAC) model application. More detailed information about data collection and ALMANAC modeling setup and results are available in Williams et al. (2017, 2020). During the initial year of establishment in the simulation, three different vegetation types were planted on March 1. The T6 measured plant density for each vegetation type was used in the model. The simulated fall

Table 1
The summer and winter irrigation operation schedule of the T6 wetland at Colusa National Wildlife Refuge in California, United States.

Year	Date	Operation	Amount (mm)
2017	May 9	Discharge water from weirs	0 to 50 depth
2017	June 9	Summer irrigation	300 to 350 ponding
2017	June 10	Store water	300 to 400 depth
2017	June 15	Discharge water from weirs	0 to 50 depth
2017	July 15	Mowing wetland plants	—
2017	Sept. 18	Fall irrigation	300 to 490 ponding
2017	Sept. 19	Store water	300 to 400 depth
2018	May 9	Discharge water from weirs	0 to 50 depth
2018	June 16	Summer irrigation	300 to 350 ponding
2018	June 17	Store water	300 to 400 depth
2018	June 28	Discharge water from weirs	0 to 50 depth
2018	July 6	Summer irrigation	300 to 350 ponding
2018	July 10	Summer irrigation	300 to 350 ponding
2018	July 11	Summer irrigation	300 to 350 ponding
2018	July 15	Mowing wetland plants	—
2018	Sept. 18	Fall irrigation	300 to 490 ponding
2018	Sept. 19	Store water	300 to 400 depth

irrigation water application was scheduled for September 18, and each wetland cell remained inundated until December 31. After the initial year of simulated establishment, the summer and winter irrigation operation cycles were repeated for the remaining period between 2015 and 2018.

After modifying APEX1501 code, the APEX model was calibrated and validated using a simple calibration/validation strategy that is the most appropriate for optimizing model performance at a specific location or areas with uniform environmental characteristics (e.g., soil, slope, and vegetation) (Daggupati et al. 2015). In the simple calibration/validation strategy, the spatial variation in biophysiochemical processes (e.g., hydrological processes, sediment transport, plant uptake, and nutrient transformations) are assumed to be minimal, and observed data collected at one location can be considered representative of the entire area (Daggupati et al. 2015). Four criteria (objective function) were used to evaluate the model's performance during calibration and validation: the percentage bias (PB), the root mean square error (RMSE), the coefficient of correlation (R^2), and Nash-Sutcliffe efficiency (NSE).

For model calibration, water level measurements under various flooding conditions collected from T6 between April of 2017 and May of 2018 were used. Then the model was validated with data collected from T6 between June of 2018 and September of 2018. During the calibration, water depth changes and the total volume of water held in T6 during summer and winter flooding periods were compared with simulation results. Since water depth data in T6:4 was not available, we compared between measured and simulated total volumes of water used for summer and fall irrigations between April of 2017 and May of 2018 in T6:1-3. The total volumes of water were estimated based on the measured and simulated water depths and area of each wetland cell. We assumed that surfaces of T6:1-3 are nearly flat and do not influence wetland hydrology such as local impoundments or diversion of flows within cells. Since no observed data are available for water discharges to wetland during irrigation operations, the water level and schedule of weir discharges were adjusted manually by comparing observed and simulated total volumes of water.

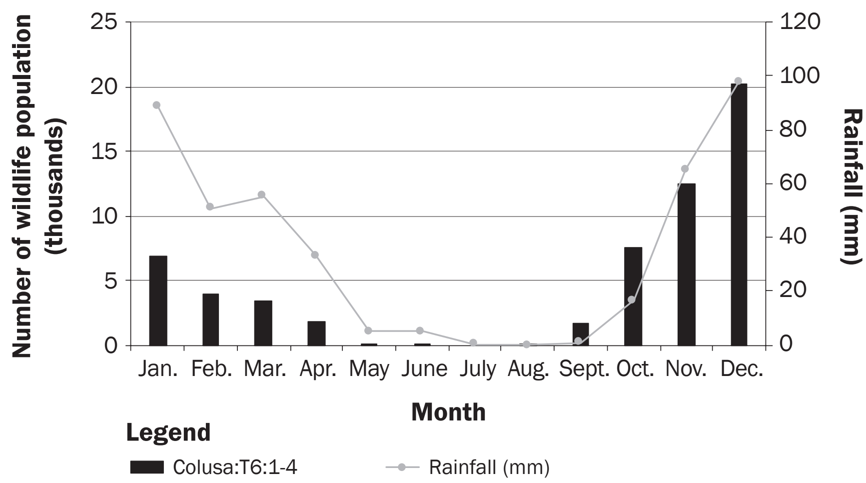
After achieving a good match (PB $\leq \pm 15\%$) between measured and simulated total volumes of water results (Moriasi et al. 2007), measured nutrient concentration in wetland water bodies collected from T6:1-4 between 2017 and 2018 was used for model calibration. Subsequently, the model was validated using data collected between 2015 and 2016. The major nutrient, N, enters and leaves the wetland in inflow and outflow water, in animal droppings, and in sediment; in addition, N movements include transfer to and from gas phases in the atmosphere (Kadlec 1979). Since T6 supports a large number of waterfowls, bird droppings may contribute the most major nutrients (e.g., N and P) entering the waterbody (Valiela et al. 1991; Scherer et al. 1995). In this study, the calculated nutrient (e.g., N) loadings by bird droppings were manually input to the model. Also, APEX was set to allow for N to enter the wetland through precipitation and agricultural drainage runoff from the 2047 Canal. The average N concentration in precipitation was set at 0.65 mg L^{-1} according to Hember (2018). However, little information was available on N concentration in agricultural runoff in the study area.

Denitrification and ammonia volatilization could be major mechanisms for N removal in wetland (Knight et al. 2000). Nitrogen loss via denitrification in flooded soil can vary by N input sources (e.g., fertilizer, urea, etc.), amount of N application, and environmental conditions (e.g., soil, climate, vegetation cover, etc.) (Kadlec 1979; Samson et al. 1990). According to Kadlec (1979) and Samson et al. (1990), who measured rates of N loss via denitrification in floodwater treated with different N applications, denitrification rate can vary from 46% to 91% of the applied N with treatments of different N source and N application rate. Also, denitrification rate increased from 61% to 87% of applied N by prohibiting plant growth in the water (Samson et al. 1990). In addition, plant nutrient uptake is also another mechanism to remove N in waterbody. According to Kadlec (1979), wetland vascular plants can uptake 2% to 10% of total N budget for a wetland, and about 50% of the N in plant biomass can be released from dead plant materials. Based on the field observation, plant density (number of plants per m^2) differed by wetland cells, so relative rates of N uptake by wetland vascular plants for T6:2 through T6:4 were calculated from the N uptake rates in T6:1. Due to factors listed above, though, the N removal data are unavailable in this study. It is expected that the N in a waterbody can be reduced between 50% and 100%. We adjusted N removal percentage in each wetland cell by comparing measured to simulated N concentrations in the waterbody. In simulation, N concentration is reduced by removing 52.3%, 51.7%, 53.1%, and 63.1% of total N amount in runoff, waterbody, and sediment in T6:1, T6:2, T6:3, and T6:4, respectively. To compare simulated and observed values, statistical measures such as the PB, RMSE, and R^2 were used.

Analytical Approach. We developed and modeled six scenarios that represent projected wetland management conditions under different combinations of two climates and four irrigation managements by utilizing the calibrated APEX model and the ARM model. California's Fourth Climate Change Assessment (Houlton and Lund 2018) reported that the Sacramento Valley will likely appear to increase daily maximum temperature by 5.56°C by 2100, while the average annual precipitation is expected to remain the same (Huang and Ullrich 2016; Pierce et al. 2018). In simulations, two climate scenarios (recent historical climate scenario, 2015 to 2018, and future climate scenario created by increasing maximum temperature of recent historical climate by 5.56°C) were run for a four-year time period. For future climate scenario, the daily maximum temperature from recent historical climate data (2015 to 2018) was increased by 5.56°C for the entire four-year time period. To compare the ARM simulated daily N contribution (dry kg ha^{-1} d^{-1}) by waterfowl between two climate scenarios, a Wilcoxon rank-sum test was performed. For summer and fall irrigation operations, T6 wetland was irrigated with four different water quantities including current water amount, increased current water amount by 1.5 and 3 times, and decreased current water amount by 0.5. In each scenario, APEX model computed total N amount stored in waterbody, N yield in runoff, and N yield in sediment for all wetland cells.

Figure 2

Monthly mean number of the birds and monthly rainfall (mm) at a managed wetland, T6, at Colusa National Wildlife Refuge (CNWR) in Colusa, California, between January of 2015 and December of 2018. The total number of birds per month and monthly rainfall were averaged over the years.



Results and Discussion

Estimated and Simulated Total Nitrogen Loading of Wetlands by Bird Droppings. Wintering waterfowl arrived in September and stayed until April at the managed wetland, T6, in 2015 to 2018 (figure 2). The highest number of birds was observed in November and December. Average bird population between September and April peaked at 20,170 birds in December, and fell to 1,825 birds in April, which included all nine major bird contributors (e.g., white fronted goose, northern pintail, white geese [snow and Ross's geese combined], green-winged teal, northern shoveler, mallard, American wigeon, American coot, and gadwall).

There was a distinct seasonal pattern in nutrient input by waterfowl according to waterfowl abundance with highest values in December and lowest input during summer. The annual mass of N entering wetland in bird droppings was estimated to be 1,112 kg y^{-1} total N averaged in 54.7 ha from 2015 to 2018 (figure 3). This is equivalent to area loadings of 2.03 g m^{-2} y^{-1}. For comparison, Manny et al. (1994) reported the average annual total N loading in Canada goose and mallard droppings over four years in the 15 ha Wintergreen Lake, Michigan, was 1.87 g m^{-2} y^{-1}.

Using month of data collection, total precipitation, and maximum temperature collected from 2015 to 2018, an ARM was developed for estimating the total N loadings (kg) to T6:1-4 wetlands by bird droppings. The ARM equation is as follows:

$$Y = 213.32 + 25.25X_1 - 11.18X_2 - 1.02 \times 10^{-8}X_3^2 + \varepsilon, \varepsilon \sim N(0,4800) \quad (2)$$

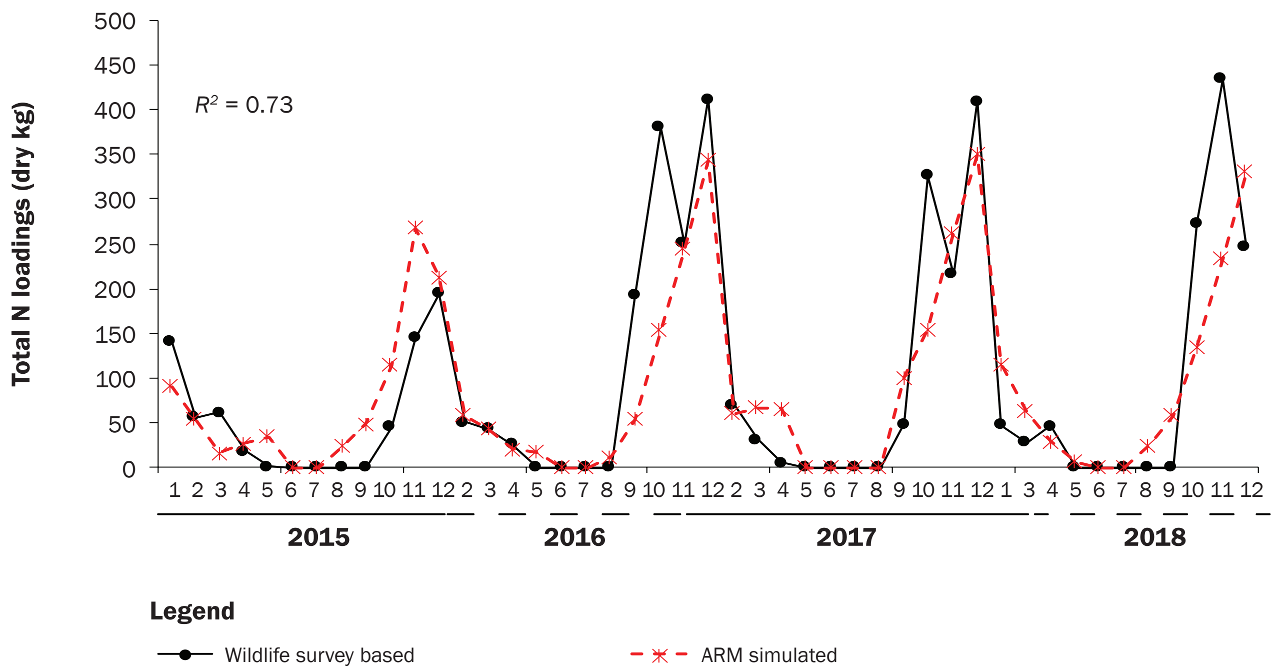
In equation 2, *p*-values of month of a year (X_1), maximum temperature (X_2), and total water amount (X_3) are 0.25 × 10^{-9}, 6.96 × 10^{-9}, and 0.023, respectively. Assuming that the smaller *p*-values are statistically significant at α = 0.05, month of a year (X_1) is the most significant variable on the potential N loadings by bird droppings. In other words, month of a year (X_1) is highly corrected with the response variable (i.e., the total N mass entering wetland in bird dropping). Like the estimated values, the simulated total N loads from September to April was over 95% of the total N entering the wetland. The maximum temperature was the second most influencing factor of N loads. Temperature changes can provide significant effects on waterfowl population numbers (Guillemain et al. 2013). According to Guillemain et al. (2013), climate changes can significantly affect migration distance, distribution, and reproductive success in ducks. Total water volume has a weak relationship with N loads. This may be because the T6 wetland is already fully inundated during winter to provide for waterfowl habitat, and additional water level has little effect on total N loads.

Overall, ARM simulated total N loadings by bird droppings agreed well with the estimated N loadings across months within a year. Regression analysis for ARM simulated and measured N loadings including all data from T6:1-4 from 2015 to 2018 revealed an R^2 of 0.73 (figure 3). The simulated yields and measured N loadings were not significantly different ($P \leq 0.0001$).

APEX Modeling Results. Between April of 2017 and September of 2018, the managed wetland, T6, was irrigated in summer to improve wetland plant seed production (Naylor 2002), which is important for foraging wintering waterfowl. The wetland remained inundated from September through April until it was drawn down, and then received a summer irrigation to bring seed-producing wetland plants to maturity to provide forage for wintering waterfowl (CVJV 2006). Figure 4 compares the measured and simulated water depth in the wetland basin of T6:1-4 during the calibration and validation periods. The partially inaccurate and missing data between April of 2017 and September of 2018 caused imperfect matches between simulated and

Figure 3

Calculated and additive regression model (ARM) simulated total nitrogen (N) loading of an artificially managed wetland, T6, at Colusa National Wildlife Refuge (CNWR), in Colusa, California, by bird droppings between January of 2015 and December of 2018.



measured water depth in T6:4 (figure 4). For evaluating model performance, the total volumes of water used for summer and fall/winter irrigations in T6:1-3 were calculated using water depth and area and compared with simulated water volumes, and the model performance was evaluated using PB, R^2, RMSE, and the NSE (table 2). Calibration results for T6:1-3 show that the simulated total water volumes in T6:1-3 are in good agreement with measured ones. As shown, the PB, RMSE, R^2, and NSE for calibration were 9.81, 6,690 m^3, 0.84, and 0.97. Based on the general performance ratings of Moriasi et al. (2007), the water volume simulations in this study may be evaluated as "very good." Unlike calibration, NSE value for validation was below the satisfactory guideline (0.5) even though the values of PB (–8.51%) and R^2 (0.71) can be evaluated as "very good" based on the general performance ratings of Moriasi et al. (2007). The low NSE may be affected by the small sample size collected only between June of 2018 and September of 2018 (McCuen et al. 2006). According to McCuen et al. (2006), the NSE becomes a better estimator as sample size increases. In simulation, the total volume of water used for irrigations in T6:1-4 between April of 2017 and September of 2018 was 581,577 m^3.

Based on measured values during the spring/summer months of May to August from 2015 to 2018, total N concentration in water increased as water flowed from T6:1 to the outlet of T6:4 (table 3). This may be because discharge water carried N to downstream cells along the downgradient direction during irrigation drawdown periods. Because of the high N concentration during the spring/summer months when wetland plants mostly grow and uptake nutrients, T6:4 had the highest vegetation covers of grasses and forbs among the four T6 wetland cells (figure 5). According to Gill et al. (2006) and Bishop et al. (2010), biomass and cover of non-N fixing plants in the herbaceous species dominated community responded rapidly to N addition. Also, N input exhibited a positive linear relationship with plant leaf morphological characteristics (e.g., leaf elongation rate, leaf appearance rate (leaf d^{-1} tiller^{-1}), and leaf length) (Costa et al. 2013). This may explain why the vegetation coverage increased significantly with increasing N availability in T6:4.

Generally, N concentration in wetland water is highly correlated with wetland hydrology (Lenhart et al. 2016). In this study, N contents in the wetland waterbody were calibrated and validated using the measured values collected during late spring/summer periods between 2015 and 2018. As shown in table 2, the model performance was acceptable in predicting N concentration in surface waterbody after refining plant N uptake and denitrification rate in T6:1-4. According to the general performance ratings of Moriasi et al. (2007), the N concentration simulations in this study may be evaluated as "satisfactory" (PB > 35% for both calibration and validation) (table 2). The values of RMSE and R^2 were 1.73 mg L^{-1} and 0.44 for calibration and 1.55 mg L^{-1} and 0.11 for validation. The NSE values for calibration and validation were below the satisfactory guideline (0.5) (table 2). The low NSE values can be affected by the limited sampling numbers from the wetlands or N input uncertainty in simulation. The values of N input by bird droppings were predicted values, which can cause either overpredicted N concentration in summer 2017 and 2018 or underpredicted N concentration of surface of waterbody in summer 2015 and 2016. This approach produced low NSE values but followed the measured increasing pattern in total N concentration in water as water flowed from T6:1 to the outlet of T6:4 (table 3).

Figure 4
Comparison of daily measured and simulated wetland water depth (mm) for calibration (April of 2017 to May of 2018) and validation (June of 2018 to August of 2018) in T6:1-4 at Colusa National Wildlife Refuge (CNWR), Colusa, California. There are missing measured water depth values for T6:4 between July of 2017 and May of 2018.

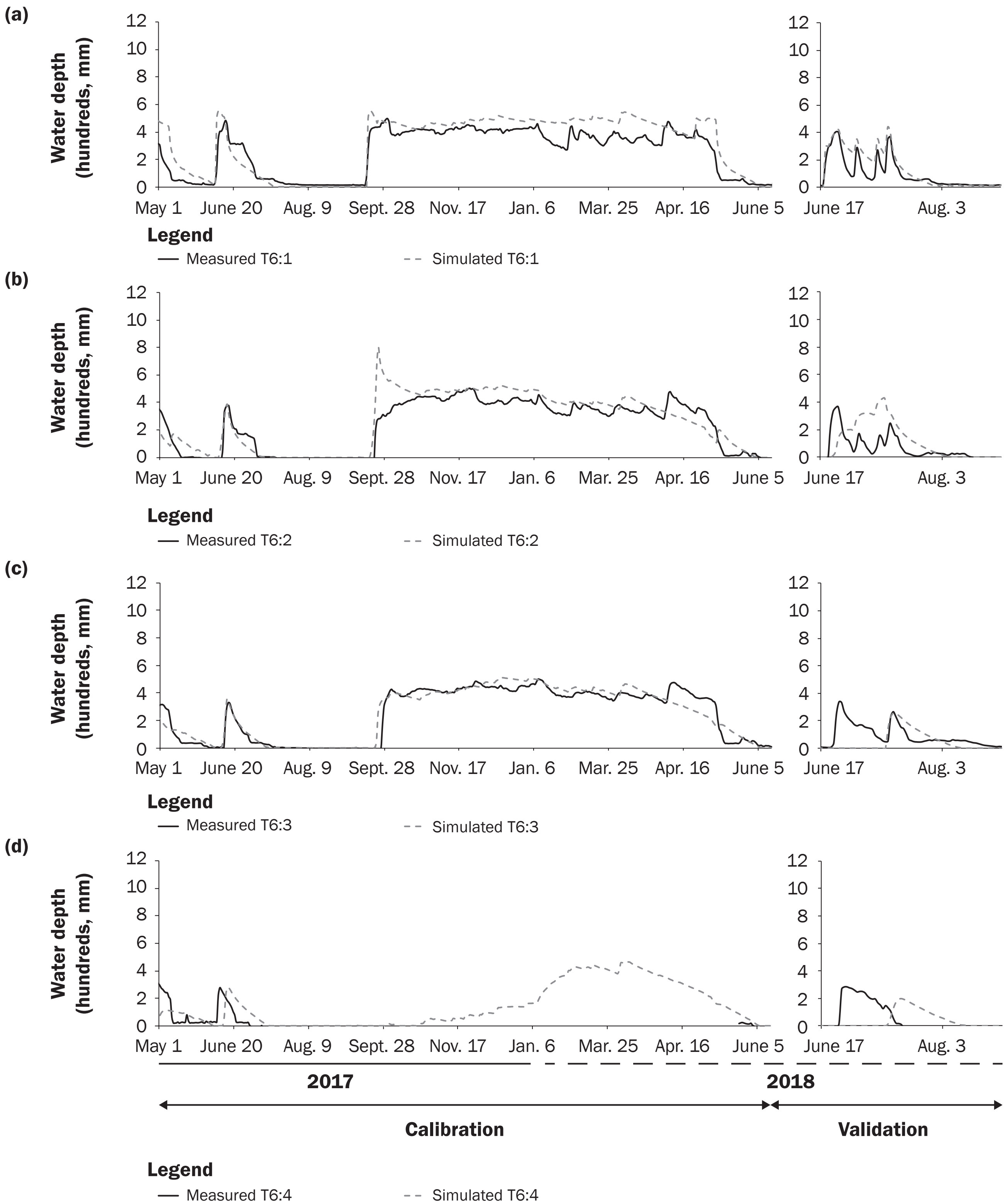


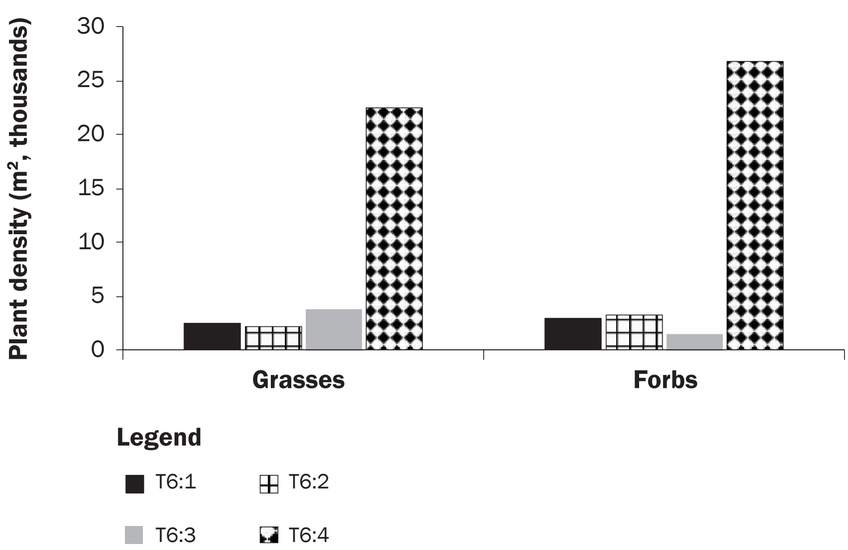
Table 2
Calibration and validation statistics for the total water volume in T6:1-3 and nitrogen (N) concentration (mg L⁻¹) in surface of waterbody in T6:1-4 at Colusa National Wildlife Refuge. PB is the percentage of bias measures; RMSE is the root mean square error; R^2 is the coefficient of correlation; and NSE is the Nash-Sutcliffe Efficiency.

Period	Observed	Simulated	PB (%)	RMSE	R^2	NSE
Mean total water held in T6:1-3 (m^3)						
Calibration (summer of 2017 to spring of 2018)	49,920	54,025	9.81	6,690	0.84	0.97
Validation (summer of 2018)	45,949	39,234	–8.51	8,754	0.71	0.27
N content (mg^{-1}) in waterbody in T6:1-4						
Calibration (summer of 2017 and summer of 2018)	1.78	2.5	39.26	1.73	0.44	–8.11
Validation (summer of 2015 and summer of 2016)	2.29	1.19	–36.59	1.55	0.11	–0.86

Table 3
Measured and simulated nitrogen (N) concentration (mg L^{-1}) in surface of waterbody in T6:1-4 at Colusa National Wildlife Refuge (including T6:1-4), late spring/summer periods between 2015 and 2018 and relative ratio between simulated and measured values.

	N content (mg L^{-1})		
Cell	Measured	Simulated	Simulated/measured
T6:1	0.89	0.80	0.90
T6:2	2.11	1.48	0.70
T6:3	2.30	2.45	1.07
T6:4	3.05	3.24	1.06

Figure 5
Mean plant density (number of plants per m^2) of two vegetation types, grasses and forbs, in T6:1-4 during late spring/summer periods in 2015.

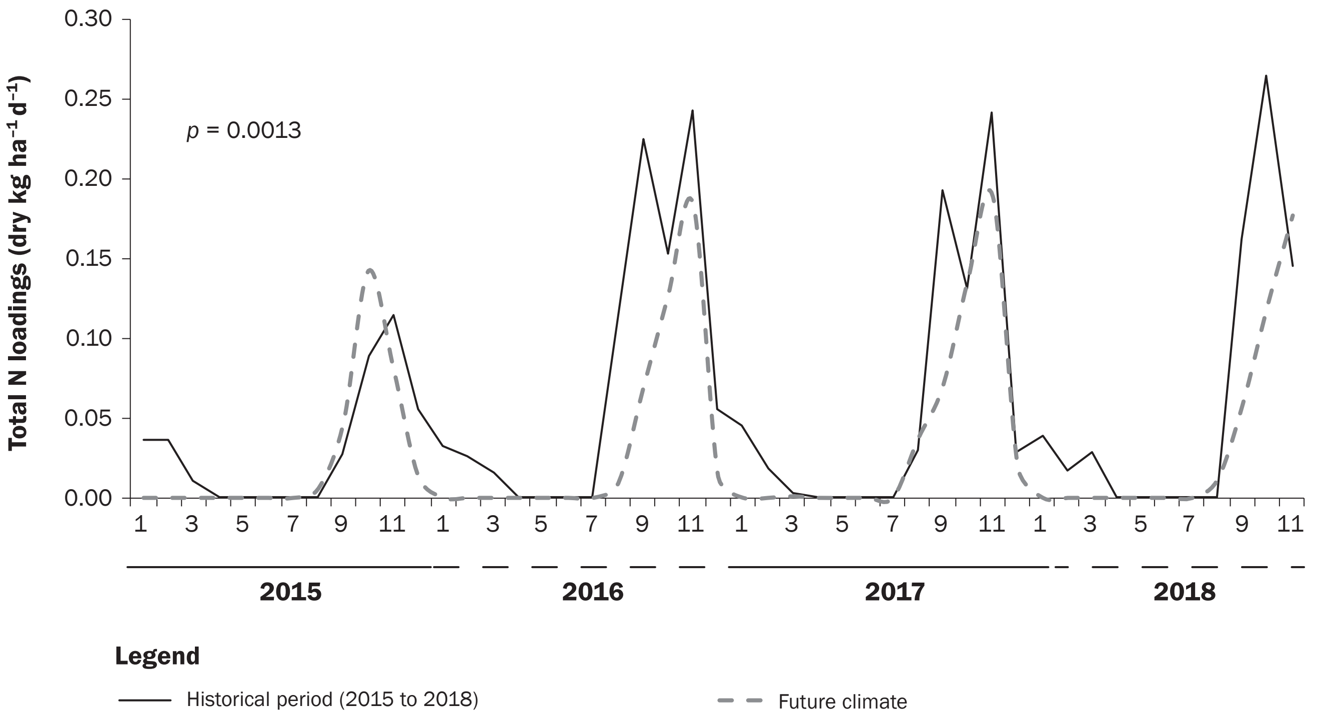


Modeling the Potential Impacts of Climates and Irrigation Managements. Projected amounts of N loadings by bird droppings were significantly decreased by up to 42% with rising temperatures (p = 0.0013, figure 6), implying that waterfowl population will be negatively influenced by temperature increases. In the historical period (2015 to 2018), there were two highest peaks for the daily mass of N entering wetland per hectare in bird droppings between fall and spring. However, in the future climate scenario, there was only one highest N contribution observed in winter time (figure 6). The simulation results appear to be well supported by Olson (2016) and Westerberg (2017) who reported that the California and the Pacific Flyways have experienced a major decline in waterfowl populations with increasing temperature and severe drought events from 2012 to 2015. In addition, previous studies have projected that climate change will significantly affect the distributions of waterfowl during fall/winter (Guillemain et al. 2013; Notaro et al. 2016; Westerberg 2017). Notaro et al. (2016) projected that fall/winter migration will be delayed for most waterfowl species, especially for mallards, with rising temperatures, resulting in at least 50% declines in waterfowl abundance in the Mississippi and Atlantic Flyways by late 21st century. According to Westerberg (2017), with continued drought conditions resulting from temperature increase and reduced water availability, along with increasing agricultural practice and urbanization, waterfowl will continue to suffer in finding suitable habitats.

The significant reduction in N loads by bird droppings significantly affects the N concentration in waterbody in wetlands (figure 7). Projected N yields (kg ha^{-1} y^{-1}) in wetlands would be cut in half under future climate warming scenarios. However, the N yields in each wetland cell in each irrigation scenario showed similar patterns between historical and future climate periods. Overall, N yields in T6:1-4 would be high during winter inundation, but N yields would be low during drought seasons (late June to August) (figure 7). T6:1-3 would generally have higher N yields in the ponding water than T6:4 under all current and irrigation scenarios 1 (increase irrigation volume by 1.5 times from current irrigation amount) and 3 (decrease irrigation volume by 0.5 times). This may be highly related to water availability in each cell. As shown in figure 4, T6:1-3 had greater water detention time than T6:4 during flooding periods, which means more time to remove nitrate (NO_3^-)

Figure 6

Additive regression model (ARM) simulated total nitrogen (N) loading rates (dry kg ha^{-1} d^{-1}) by waterfowls in T6:1-4 at Colusa National Wildlife Refuge during historical reference scenario (2015 to 2018) and climate scenario (historical Tmax + 5.56°C in 2015 to 2018). The Wilcoxon rank-sum test was performed to compare average total N contribution between historical reference and climate scenarios at α = 0.05.



and organic N from the ponding water in these cells.

In scenario 2 (increased irrigation water volume by 3 times), N yields in surface waterbody in T6:1 and T6:2 would be lower than current and scenarios 1 and 3 during winter/spring flooding period (figure 7) due to dilution effect of increased inflow volume, though N concentration in downstream cells were less influenced by the increased irrigation volume. In general, APEX results indicate that the system of wetlands represented by the four cells will perform consistently to remove N efficiently as evidenced by the lowest N yields at the final outlet (T6:4) in all scenarios in each cell as water flows from T6:1 through the outlet of T6:4.

Table 4 presents the simulated results for the highest peak daily QN discharge from the outlet of T6:4 after winter/spring flooding period. Due to climate change impacts (e.g., lower N loadings by bird droppings), increased temperatures led to lower QN yields in discharge water in all scenarios. However, QN yields in discharge water increased in scenarios 1 and 2 with greater water irrigation volumes. In scenario 3, when water irrigation volume decreased, QN yields in discharge water would be very low with a range between <0.005 and 0.04. According to the simulated results, the N removal efficiency by the wetland may decrease as the volume of irrigation water increases because excessive N can be carried in runoff by increasing discharge volume. Similar results were observed by Cui et al. (2016) who reported low N removal rates by wetland during summer because of increasing hydraulic loading rates derived from high rainfall. In addition, Li et al. (2018) reported that irrigation volume had more significant effects on leaching water quality than N deposition. Under the same N deposition, excessive irrigation strongly increased nutrient removals and influenced pH, electroconductivity (EC), and concentrations of P, potassium (K), calcium (Ca), magnesium (Mg), sodium (Na), and copper (Cu) (Li et al. 2018).

However, according to Colusa NWR wetlands management plan (US FWS 2016), wetlands must be flooded to a certain depth to function within their system. To move water from one cell to the next, each cell must reach a certain depth to be able to continue to flow to the downstream cells. In addition, reducing the amount of irrigation water may negatively affect the quality of existing waterfowl habitat by reducing foraging depth at the peak of fall waterfowl migration. Isola et al. (2000) studied habitat use by waterfowl foraging in managed, seasonal wetlands in northern San Joaquin Valley, California, and reported that the optimal water depths at foraging sites varied with bird body size. Four bird groups including small shorebirds, large shorebirds, teal, and large dabbling ducks were mostly found at depths of <5, 5 to 11, 10 to 15, and >20 cm, respectively (Isola et al. 2000). Further carefully designed studies considering alternative ways to improve irrigation efficiency without affecting the quality of waterfowl habitats are needed. For example, effects of vegetative filter strips and grouping plant species with similar water requirement (Bilderback 2002) on water velocity and nutrient transport in runoff can be studied.

Assumptions and Limitations. Since the EPIC plant growth model is the basis for both APEX and ALMANAC (Williams et al. 2008), we assumed that plant param-

Figure 7
Mean simulated nitrogen (N) yields (ka ha⁻¹) in waterbody in T6:1-4 at Colusa National Wildlife Refuge, of current irrigation management and three scenarios during (a, c, e, and g) historical (2015 to 2018) and (b, d, f, and h) future periods (historical Tmax + 5.56°C). In scenarios 1 and 2, the current irrigation volume was increased by 1.5 and 3 times, respectively. In scenario 3, the current irrigation volume was decreased by 0.5 times.

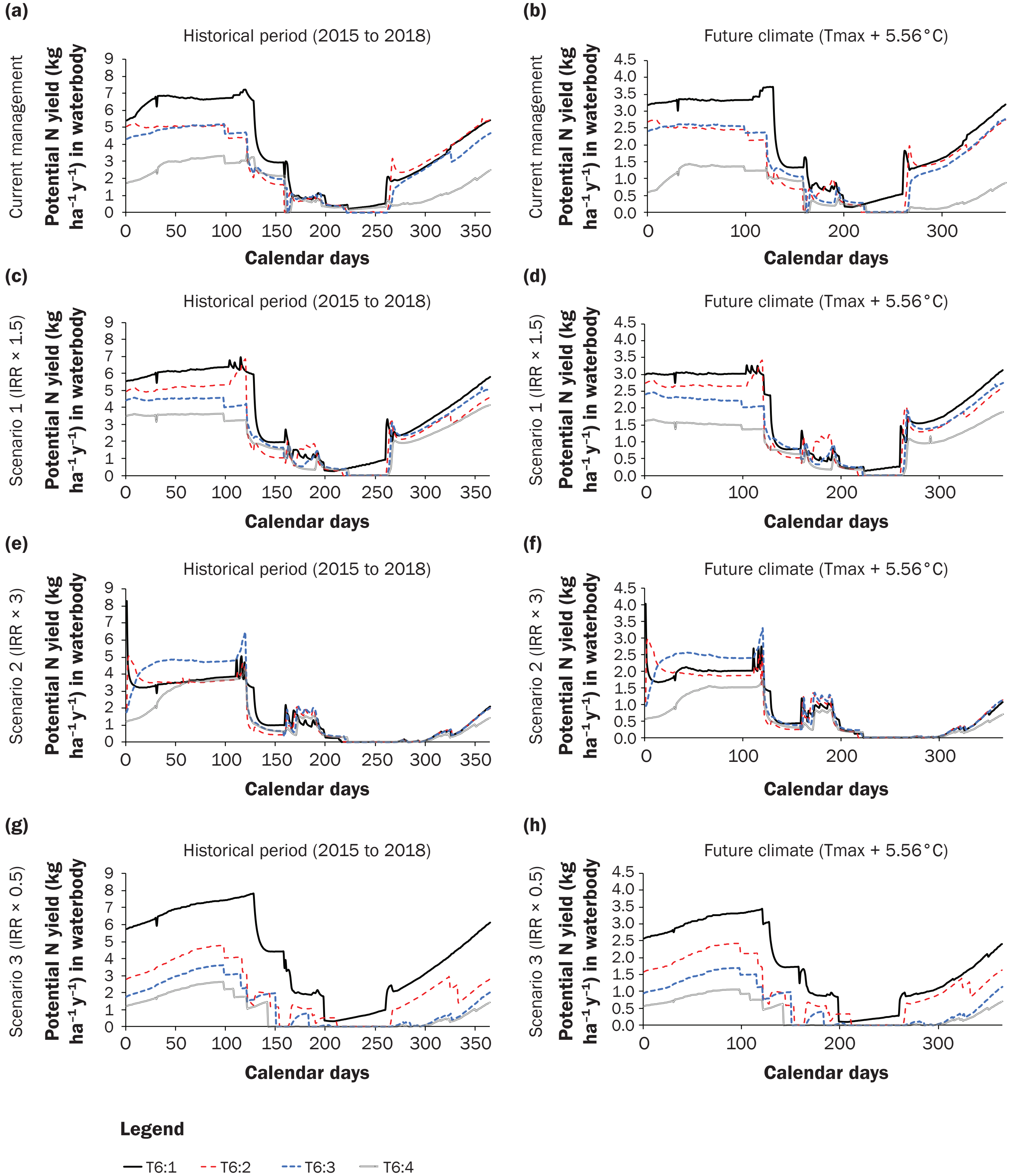


Table 4
Mean simulated the highest peak dissolved nitrogen loads in water discharge (QN) from surface water runoff from outlet at T6.4 at Colusa National Wildlife Refuge, after winter/spring flooding periods of current irrigation management and three scenarios during historical (2015 to 2018) and future periods (historical Tmax + 5.56°C). In scenarios 1 and 2, current irrigation volumes were increased by 1.5 and 3 times, respectively. In scenario 3, the current irrigation volume was decreased by 0.5 times. IRR indicates water amount used for wetland irrigation.

Date	Current management (kg ha^{-1})	Scenario 1 (IRR × 1.5) (kg ha^{-1})	Scenario 2 (IRR × 3) (kg ha^{-1})	Scenario 3 (IRR × 0.5) (kg ha^{-1})
Historical periods (2015 to 2018)				
April	0.52	0.9	1.98	0.03
June	0.35	2.17	3.92	0.00
Future climate (Tmax + 5.56 °C)				
April	0.23	0.37	0.86	0.04
June	0.17	0.47	2.04	0.00

eters for the three vegetation types (e.g., grasses and forb) and plant growth properties derived from ALMANAC model application (Williams et al. 2020) were transferable to the APEX model without losing simulation accuracy. This multimodel combination approach has been proven successful in earlier studies (Kim et al. 2018a, 2018b). Due to limited wetland access, sampling water quality data were not available during breeding and bird migration seasons. This may have led to an uncertainty in model predictions, which can be resolved by collecting detailed field data before and after winter flooding seasons in future studies. Since the wetlands were managed by farmers, no water flow data during irrigation or discharging periods were available in this study, which may also have led to an uncertainty in model prediction. This can be resolved by collecting water depth for longer time periods in future studies, which will provide enough amount of data for model calibration and validation.

Summary and Conclusions

Two models, ARM and APEX, were used to evaluate the impacts of N loading from bird droppings on water quality in a managed, seasonal wetland in Colusa, California. After successful model calibration and validation, a total of six scenarios were developed and evaluated for water quality and nutrient removal by wetlands due to co-occurring agents acting on the system: climate change and changes to water irrigation management. Based on the field observations, there was a seasonal variation in N loadings in wetlands from bird droppings. Using weather variables (e.g., temperature), total water volume, and month of a year, an ARM was developed to predict the possible N loading values. The ARM successfully predicted the seasonal pattern with peak N loadings in December, when bird abundances was greatest. The calibrated ARM evaluated the impacts of climate change on the N loadings in wetlands. By increasing temperatures, this impact would result in significant reduction in N loadings by bird droppings. The simulated N loading values were used to feed N input in the calibrated APEX model for climate change scenarios, while the N loading values estimated based on the field observations were used in the model scenarios under historical periods (2015 to 2018). Under both historical and future climate conditions, water quality and N removal efficiency were highly associated with the volume of irrigation water. Excessive irrigation strongly influenced the quality of stored water in wetlands and increased the amount of dissolved N in runoff. However, irrigation is crucial for increasing wetland plant production and providing high quality habitat. Thus, additional research is needed on increasing irrigation efficiency (e.g., development of vegetative strips) and improving nutrient removal efficiency without affecting the quality of waterfowl habitats to further inform wetland management planning. Water scarcity coupled with increasing demands for water in cities and farmlands will continue to be a growing challenge in California. Novel proposals to increase the use of wastewater to meet water demand in wetlands while improving the quality of water required to sustain ecosystem health will continue to be a common research theme in the future. Our modeling approach can be successfully used to evaluate likely effects on the quantity and quality of water in wetlands of proposed or actual changes to irrigation management (e.g., irrigation volume, irrigation cycle, etc.), vegetation management (e.g., mowing, disking, and grazing), alternative conservation strategies (water retention time and water flow velocity), and updated projections of climate changes.

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