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Conservation Reserve Program (CRP) lands provide ecosystem service benefits that exceed land rental payment costs ☆

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ABSTRACT

Global demand for commodities prompted the expansion of row crop agriculture in the Upper Midwest, USA with unknown consequences for multiple ecosystem services. The Conservation Reserve Program (CRP) was designed to protect these services by paying farmers to retire environmentally sensitive land. Here we assessed whether the benefits provided by CRP's targeted retirement of agricultural land are equal to or greater in value than the cost of rental payments to farmers. We quantified the benefits of CRP lands for reducing flood damages, improving water quality and air-quality, and contributing to greenhouse gas mitigation in the Indian Creek watershed in Iowa. We found that for all assessed scenarios of CRP implementation, the ecosystem service benefits provided by CRP lands exceed the cost of payments to farmers. Expanding CRP implementation under one of three potential scenarios would require an average per-acre payment of $1311 over the life of a 10-year contract but would generate benefits with a net present value of between $1710 and $6401. This analysis suggests that investment in CRP in Indian Creek, and likely in other watersheds in the Upper Midwest, is justified based upon the value of public and private benefits provided by CRP lands.

1. Introduction

Cultivation of row crops in the Midwest US expanded as demand primarily for biofuel drove rising commodity prices and prompted further conversion of perennial land cover (Johnston, 2013; Wright and Wimberly, 2013; Lark et al., 2015). This conversion typically increases nutrient and sediment loading and degrades water quality (Donner and Kucharik, 2008; Secchi et al., 2011). Best-management practices and targeted conservation of lands can help mitigate these impacts and provide valuable ecosystem services (Schulte et al., 2006; Swinton et al., 2007; Asbjornsen et al., 2014) and as a result conservation is an important part of US Federal Farm programs.

The Conservation Reserve Program (CRP) has been a critical component of agricultural conservation since its creation in 1985 and supports voluntary retirement of "environmentally sensitive" lands in agricultural production (FSA, 2015). The CRP program provides annual rental payments, cost share assistance, and in some cases additional incentives to farmers who restore enrolled lands for the duration of 10-or 15-year contracts (Stubbs, 2014). The program has been implemented extensively across the US with 24.3 million acres (9.8 million ha) currently enrolled, down from a peak of approximately 37 million acres (15 million ha) in 2007 (FSA, 2015). Research has shown that CRP lands provide a variety of environmental benefits including, for example, creating wildlife habitat (Drum et al., 2015), reducing soil erosion (Sullivan et al., 2004), restoring hydrology and reducing nutrient loading (Gleason et al., 2011), enhancing groundwater recharge (Rao and Yang, 2010) and storing carbon and reducing greenhouse gas emissions (Gelfand et al., 2011). Despite evidence of the benefits from CRP lands, support for the CRP program is declining and recent federal legislation reduced the acreage cap for the program setting a maximum of 24 million total acres that can be enrolled (Stubbs, 2014). In addition, high crop prices have prompted many farmers to not reenroll when their contracts expire, resulting in millions of acres voluntarily exiting the program (Chen and

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Khanna, 2014). There is a need for rigorous analysis of the value of benefits provided by CRP lands relative to costs of implementation. Previous research using national data and modeling approaches has estimated the economic value of enhanced recreational opportunities provided by CRP (Feather et al., 1999) and calculated the total annual value of benefits provided by all CRP acres (Hansen, 2007). However, there has been no assessment of CRP that links quantification of specific biophysical changes to economic valuation of benefits. To address this gap, we derived estimates of economic value from modeled biophysical changes to existing CRP acreage in a watershed in Iowa, and then compared the total estimated value of ecosystem service benefits to the costs of rental payments for these CRP lands.

2. Methods

2.1. Study site and land use scenarios

We valued multiple ecosystem services associated with CRP in the target watershed of Indian Creek, a 10-digit Hydrologic Unit Code (HUC) watershed covering approximately 60,000 acres (24,281 ha) located within Linn County and Eastern Iowa's Lower Cedar Basin (Fig. 1). Agricultural land use predominates, with 30,000 acres in corn and soybeans (12,141 ha) (CDL, 2012), though the watershed also contains significant urban land use including the cities of Marion and Alburnett, and portions of Robins, Hiawatha, and Cedar Rapids, and several smaller townships. The main surface water bodies are Indian Creek, its tributaries, and a number of small lakes. Indian Creek flows into the Cedar River downstream of the City of Cedar Rapids, which then joins with the Iowa River and ultimately flows into the Mississippi River.

We assessed the costs and benefits associated with varying CRP acreage by comparing the value of ecosystem services provided by acres enrolled in CRP in 2012 with the value of services provided by two scenarios of reduced CRP acreage and three scenarios of increased CRP acreage (Table S1). Baseline land use information showing the extent and location of 966 acres (391 ha) of CRP contracts in 2012 were provided under a data sharing agreement by the USDA-Farm Service Agency (USDA-FSA). The five alternative scenarios were created by Smith et al. (2013) using GIS to depict varying amounts and locations of CRP acres. A 'Partial Loss' scenario showed a landscape where grass-based CRP acres are removed from the program and converted to row crop agriculture, reducing total CRP acreage to 70 acres (28 ha) of forest cover. A 'Total Loss' scenario removed all enrolled acres from the CRP program, increasing land in corn and soy production to 30,750 acres (12,444 ha). Smith et al. (2013) also developed three scenarios of hypothetical increased CRP coverage. A 'Targeted Riparian' scenario restored all row crop lands to grass within a 30 m buffer of streams in the watershed, increasing CRP to 2374 total acres (961 ha). A 'Targeted Wetland' scenario restored to wetland any landscape sink areas larger than 1 acre (.4 ha) previously identified in an analysis completed by the Iowa DNR. Finally, a 'Combined Wetland and Riparian' scenario was also developed, integrating areas targeted by each of the previous scenarios and resulting in a total of 3922 acres (1587 ha) in enrolled in CRP or 6.5% of the watershed. Across all scenarios we assumed that enrolled acres to be in compliance with the program requirements and we assessed ecosystem services changes accordingly.

We calculated the costs of CRP for each scenario by averaging, according to soil type, records of actual payments made to producers in this watershed in 2012. Although a change in enrolled acres could be accompanied by a shift in per-acre CRP rental rates, rates in Linn County Iowa have not changed significantly in recent years despite declining enrollment (FSA, 2015). Consequently we applied the average 2012 rental rate for forest, wetland and grassland restoration practices (Smith et al., 2013) to all scenarios. This estimate is not a proxy for land value, does not include all of the administrative costs associated with managing the CRP program and also does not account for the potential social cost of public subsidies (e.g. Dahlby, 2008). However, this approach does allow use of local data about actual CRP payments to estimate how the cost of payments to farmers might change with increasing or decreasing acreages of CRP.

We compared the costs of payments to farmers with the value of select ecosystem service benefits provided CRP. We quantified the flooding, nutrient and sediment loading, and air and greenhouse gas emissions associated with each of the CRP scenarios and assessed how the costs of implementation and the value of benefits changed with increasing or decreasing amounts of CRP land in Indian Creek. To account for the potential for increased enrollment in CRP in Indian Creek to increase cultivation in other areas thereby reducing the benefits attributable to expanded CRP, we incorporated an estimate of leakage from a general equilibrium analysis by Taheripour (2006). Although it is uncertain whether new lands cultivated as a result of leakage would generate greater or lesser ecosystem service benefits, following Taheripour we assumed 20% leakage from increased enrollment in CRP, and reduced by 20% our estimate of the net value of ecosystem services provided by each scenario of increased CRP.

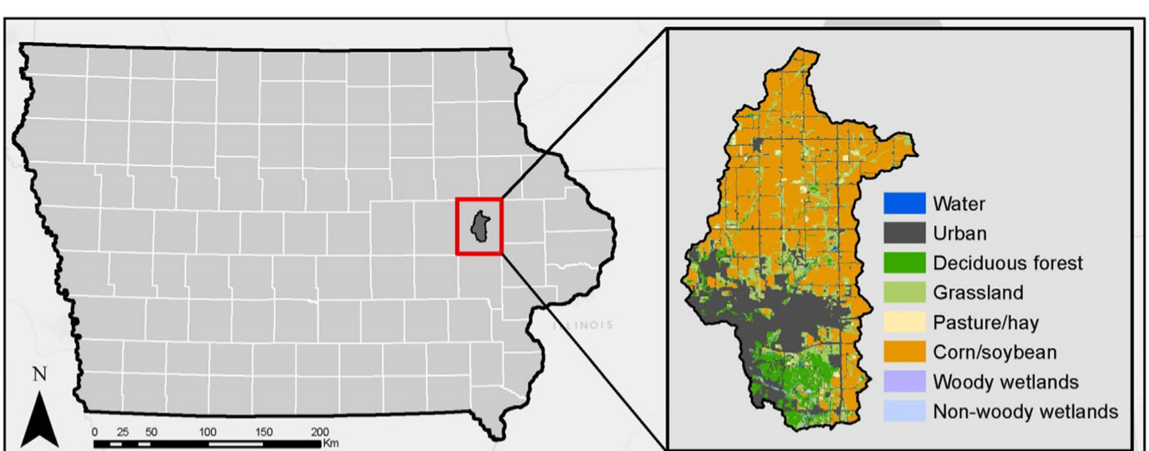

Fig. 1. *Indian Creek watershed, Iowa, USA.* Location and major land use-land cover categories of the Indian Creek Watershed, Iowa, USA.

2.2. Flood inundation and flood damage reduction

Analysis of the impact of CRP lands on flooding and associated flood damages was completed by the U.S. Army Corps of Engineers (USACE) Mississippi Valley Division (Smith et al., 2013). In 2012 and 2013, the Corps and USDA-FSA partnered to explore the flood-damage reduction benefits that upstream CRP lands provide to downstream urban areas in the Indian Creek watershed. Direct damages to structures represent only a portion of the social cost of flooding and calculating avoided damages is an incomplete assessment of the value of potential flood reduction benefits from CRP (e.g. Shabman, 1997). However, tracking changes in damages associated with flood management is the standard appraoch employed by the USACE and this method enables a rigorous comparison of how this portion of the overall costs of flooding will change under different management alternatives (U.S. Army Corps of Engineers, 2000, 3–16). To simulate the hydrologic impacts of changing amounts and locations of CRP lands the Corps utilized the Gridded Surface Subsurface Hydrologic Assessment (GSSHA) model. GSSHA is a physically-based distributed hydrologic model that simulates 2-dimensional overland flow and groundwater, 1-dimensional channel flow and infiltration, and has a full coupling between surface and subsurface components. Smith et al. (2013) used a 100 m grid resolution GSSHA model to simulate the flowrate and resulting flood stage across a range of meteorological events and for different CRP scenarios. To assess the flood damages associated with each scenario, Smith et al. (2013) applied GSSHA model outputs to LIDAR elevation data and a structure inventory using depth-damage functions from the Institute for Water Resources (Davis and Skaggs, 1992). Water surface elevations derived from GSSHA model simulations were applied to the structure inventory to identify inundated and damaged assets associated with each scenario, which is standard Corps practice for flood risk management studies (U.S. Army Corps of Engineers, 2000, 3–16). Using this approach Smith et al. (2013) estimated flood damages associated with specific rainfall and flood probabilities.

Since large and damaging flood events occur relatively infrequently we generated annualized estimates of damages associated with specific event probabilities to estimate the value of CRP for mitigating damages associated with flooding. We used USGS regression equations for peak discharge exceedance to compute the current-condition flow frequency curve at each point evaluated for impacts to structures, which was extrapolated using a log-linear function. The GSSHA model outputs associated with specific rainfall scenarios and the resulting flood events were plotted on the computed log-linear flow frequency function for the baseline and five alternative scenarios. Holding the probability of the precipitation events constant (as computed from the initial curve), a new log-linear flow frequency function was plotted for each land use scenario, connecting the points created with the new flow results from the model. The result is six log-linear flow frequency relationships, the first of which was derived from the current land use condition model results, with the five CRP scenarios generating other relationships that approximately represent the effect of altering land use on the flow frequency curve (Fig. S5). We then used the hydraulic rating table derived from a hydraulic model for Indian Creek to convert discharge at a point to its corresponding water surface elevation (stage), as the frequency analysis is based on the probability distribution of discharge. The crosswalk from flow to stage to damage is necessary because the probability basis is for flows, but the damage basis is estimated from water surface elevation.

The water surface elevations corresponding to a range of flood frequencies were applied to the inventory containing 901 structures used in the analysis. The structures were grouped into Index Points (1–28) based on proximity and all structures within an Index Point received the same water surface elevation value when determining flood damages (if any). A windshield survey was conducated to obtain first floor elevations (FFE) for all 901 of the structures used in the analysis in order to take the ground elevation (obtained using Lidar) of the structure, add the height to the first floor, and (if necessary) subtract 8 feet (2.4 m) in the case of a basement. This "zero" damage point was then compared to the peak water surface elevation output at that Index Point in order to determine depth of flooding. Next, the depth (if >0) was compared to the depth damage functions (IWR-92-R-3) and a corresponding % of structure value damage (based on structure type). The current building value was obtained from county tax assessor data, multiplied by 1.5 to include a rough estimate of content value (the Corps generally considers 50% of the value of the structure to be an acceptable surrogate for a full survey) and then further multiplied by the depth damage % to get a total damage value.

$$\textit{Flood Depth} = \textit{WSEIP} - (\textit{Elevground} + \textit{Heightfirst floor})$$
$$\text{No basement}$$

$$\textit{Flood Depth} = \textit{WSEIP} - (\textit{Elevground} + \textit{Heightfirst floor} - 8)$$
$$\text{Basement}$$

$$\textit{Percent damage to structure} = f(\textit{Flood Depth})$$

$$\textit{Damage to structure} = (\textit{Percent damage} * \textit{Assessed value} * 1.5)$$

The resulting damage frequency curves were then used to determine annualized damages for each index point and scenario. This analysis simulated how changes in CRP acreage would affect flow, inundation and flood damages, and consequently enabled calculation of the estimated annual damages associated with each CRP scenario.

2.3. Biophysical modeling of additional ecosystem services

We also evaluated the impact of the different CRP scenarios on phosphorus, nitrogen and sediment loading, emissions from fertilizer application, and greenhouse gas emissions and carbon sequestration.

To assess the impacts of CRP lands on nutrient and sediment loading we used the Soil and Water Assessment Tool (SWAT), a process-based model that was developed primarily for use in watersheds with agricultural land use (Arnold and Fohrer, 2005; Arnold et al., 1998; Gassman et al., 2007). SWAT 2015 (Rev. 637) was employed for this study. The model operates on a daily timestep and is able to simulate a wide range of agricultural management practices to determine the contribution of water, sediment, and nutrients from the landscape. Spatial data requirements include land cover, soils, and slope. Weather inputs include daily values of precipitation, temperature, wind speed, relative humidity, and solar radiation. Precipitation data were taken from gauges located in Cedar Rapids and Marion, IA and data for the remaining weather inputs were obtained from the Climate Forecast System Reanalysis (CFSR) global meteorological dataset (Fuka et al., 2014) and formatted for the SWAT model by the SWAT global weather data download website (http://globalweather.tamu.edu/). Flow data were obtained from USGS gauge number 05464695 at Marion, IA and daily flow values are available beginning in May 2012 (http://waterdata.usgs.gov/nwis/sw). Daily mean flow values were compared against model-predicted flow data in order to assess model performance. Input from the Iowa Soybean Association, Iowa State University Extension and Outreach (2015), and prior experience (Dalzell et al., 2012) informed parameterization of planting/harvest dates and tillage and fertilizer practices necessary to simulate field management and crop growth. The dominant cropping practice is a corn-soybean 2-year rotation as determined from county-level data provided online by the National Agricultural Statistics Service (http://www.nass.usda.gov/). Rotations were staggered among watershed sub-basins such that roughly half of the watershed was in the corn phase while the other half was in the soybean phase during any given year. Given the flat topography in the study area and prevalence of subsurface tile drainage systems in Midwestern row crop agriculture, subsurface tile drainage was simulated for all agricultural land situated on slopes less than 3%. The resulting estimate for the locations of tile drainage in Indian Creek watershed were very similar to estimates generated by the Iowa Department of Natural Resources (2008) which are based on slope class as well as soil drainage characteristics.

The functional unit of the SWAT model is the hydrologic response unit (HRU), which represents each unique combination of land use, soil type, and slope class. Soils data were based on the Soil Survey Geographic Database (SSURGO), slope information was based on a 30 m digital elevation model (DEM) from the National Elevation Dataset and land use/land cover information was based on the 2008 Crop Data Layer (http://www.nass.usda.gov/research/Cropland/SARS1a.htm). For the baseline scenario, the final model was comprised of 985 HRUs. Model performance is evaluated by looking at agreement between observed and predicted mean values, general correlation, and Nash Sutcliffe Efficiency (NSE) (Nash and Sutcliffe, 1970).

$$NSE = 1 - \frac{\left(Y_o - Y_m\right)^2}{\left(Y_o - \bar{Y}_o\right)^2}$$

where Y_o is the observed monthly value (discharge or load), Y_m is the modeled value of the same parameter, and $\bar{Y}_o$ is the mean value of the observed data. NSE values can range from $-\infty$ to 1. Perfect agreement between predicted and observed data results in NSE $=1$; an NSE value of 0 indicates that the model predictions are capturing the mean of the observed data. For watershed scale modeling, monthly NSE values of 0.36–0.50 are generally considered fair, values from 0.50 to 0.75 are considered good, while values greater than 0.75 indicate very good model performance (Moriasi et al., 2007; Motovilov et al., 1999). The model was calibrated over the 13 month period from 1 June 2012–30 June 2013 and validated over the 13 month period from 1 July 2013–31 July 2014) This calibration and validation period reflects the time of overlap between available stream flow monitoring data and weather input data needed to run the model. Monthly NSE values were 0.90 and 0.92 for model calibration and validation, respectively, indicting very good model performance (SI Fig. S2).

There were no monitoring data for sediment or nutrient export from the study watershed which precludes site-specific calibration for these water quality parameters. However, nutrient calibration has been performed in other published studies of the larger Cedar River basin that employed the SWAT model (Wu et al., 2013) and a nutrient calibration parameter value published in that study (NPERCO) has been applied to this model. Area-normalized NO_3^- and mineral phosphorus export from the baseline model of the Indian Creek watershed were 15.0 and 0.44 kg ha^{-1}, respectively. These values are comparable to area-normalized loads observed throughout the Cedar River basin as well as modeled loads for the region encompassing Indian Creek presented in Wu and Liu (2012).

Following calibration and validation of the SWAT model, we applied the alternative land management scenarios developed by Smith et al. (2013). To simulate transition from cropland to CRP, the corn-soybean rotation was replaced with a perennial prairie grass (and vice versa for simulating loss of CRP). For scenarios with buffer strips, vegetation change from cropland to perennial grasses was accompanied by simulation of edge-of-field filter strips for managed croplands (filter strip width = 10 m, Lee et al., 2004). Riparian wetland land cover classes included an increase in surface roughness for the wetland as well as increasing channel roughness in tributary channels. Increasing roughness has the effect of slowing overland runoff and channel flow, which increases settling of sediment and associated phosphorus and also increases the opportunity for in-channel transformations of nitrogen. A summary of model changes for alternative scenarios is presented in Table S3.

Model results were evaluated by looking at average annual sediment and nutrient loading from the entire watershed and compared against the baseline condition. It should be noted that this approach assumes that there are no important non-field sources of sediment (such as failing stream banks) in Indian Creek and that buffer strips function in a manner representative of the primary studies that were used to establish these relationships in the SWAT model (Neitsch et al., 2011; Muñoz-Carpena et al., 1999).

2.4. Valuation of additional ecosystem services

Modeled changes in air quality, carbon sequestration, and water quality have different impacts on well-being that require tailored valuation approaches. Changes in sediments and nutrients can alter ground- and surface water quality and subsequently impact recreation, property values, and treatment costs, among other hydrologic services (Brauman et al., 2007; Keeler et al., 2012; Olmstead, 2010). Connecting biophysical changes in water quality to human well-being and assessing the value of changes in the services provided is complex and context-dependent (Keeler et al., 2012; Griffiths et al., 2012). For sediment and P, we link changes in these constituents to surface water quality and corresponding effects on water use and non-use values, such as recreation, home values, and other surface water-related activities. For N, we link changes in nitrate loading to impacts on drinking water and estimate avoided costs that private well-owners may pay to avoid health risks associated with exposure to elevated N concentrations in groundwater. We also estimated the potential future costs incurred by public water suppliers. Detailed methods for each valuation approach are described below. All values are expressed in 2013$ adjusted using an average annual CPI (Bureau of Labor Statistics, 2015). We calculated net present values for each CRP scenario assuming a 10-year CRP contract and a discount rate of 3%, which is consistent with the US Office of Management and Budget's social rate of time preference.

2.4.1. Sediment and surface water quality

Changes in sediment loads affect the provision of ecosystem services such as recreational fishing, boating and swimming, navigation, and drinking water supply. We applied a methodology that integrates the value of sediment retention across fourteen benefit categories published by the USDA Economic Research Service (ERS) (http://www.ers.usda.gov) (Hansen and Ribaudo, 2008). Hansen and Ribaudo (2008) provide estimates of the marginal economic value of changes in sediment loads for HUC-8 watersheds expressed as willingness-to-pay (WTP) for reduced soil erosion. These per unit sediment costs are derived from studies using a variety of valuation approaches including costs associated with dredging, lost recreational value, municipal and industrial water treatment, and soil productivity and are then prorated to different watersheds and multi-state farm production regions based on the types of benefits and published cost estimates appropriate to those areas (Hansen and Ribaudo, 2008). We applied these WTP estimates from Hansen and Ribaudo (2008) for the Lower Cedar sub-basin that contains Indian Creek to derive an inflation-adjusted WTP benefit of $6.37 per ton change in sediment. This approach is consistent with valuation estimates for sediment reduction used in other agricultural ecosystem service assessments at the watershed scale in the Corn Belt region (e.g., Dalzell et al., 2012; Gascoigne et al., 2011; Meehan et al., 2013). To apply this WTP to modeled sediment loads in each scenario and to allow decreasing returns with greater reductions in sediment loading, we prorated the value assuming a nonlinear relationship based on a power function of the form $f(x)=WTP=\alpha^{*}x^{1/2}$, where the constant scaling coefficient α (48,097) was calculated to match the WTP for a 50% reduction in sediment loading from baseline. We estimated that adding CRP would provide sediment retention services with a net present value between $107,702 and $192,426.

2.4.2. Phosphorus and surface water quality

CRP expansion scenarios were associated with 17% to 61% reductions in modeled phosphorus loading (Table S3). Reduced P export can enhance the provision of multiple ecosystem services including fish production, recreation, and property values (Brauman et al., 2007). These benefits may be derived by residents living in the watershed or those who visit affected water features. To estimate P-related benefits associated with each CRP scenario, we applied a benefits transfer approach developed by Johnston et al. (2005).

Johnston et al. (2005) conducted a national meta-analysis to estimate annual average per household values for improvements in surface water quality. Johnston et al. (2005) used this meta-analysis to develop a benefits transfer function that estimates WTP for water quality changes based on deviations from a defined water quality baseline measured by the Resources for the Future water quality ladder (Carson and Mitchell, 1993). The water quality ladder links biophysical characteristics (e.g., water clarity, dissolved oxygen, pH) to changes in the suitability of surface water for different uses (such as boating, swimming, and fishing). The benefits transfer function has a general model form of $\ln(WTP)=intercept+\Sigma(coefficient_i)(assigned\ variable\ value_i)$, where the coefficients and independent variables are derived from the meta-analysis and adjusted by the user for a given application and context (Johnston and Besedin, 2009).We parameterized the function with information on median household income ($48,272 per year for Linn County, IA based on and inflation-adjusted from 2000 US Census data), region (plains state), and surface water type (single river) based on characteristics of the Indian Creek Watershed (Johnston and Besedin, 2009).

The benefits function requires users to specify a baseline water quality. Changes in benefits are then estimated relative to this baseline to account for non-linearity in marginal values of clean water (e.g. watersheds with a very high baseline water quality will generate small returns to improvements in quality whereas poor quality watersheds are more likely to have greater marginal returns). To estimate baseline water quality we used impairments and stressors data from the Iowa Department of Natural Resources for Indian Creek and its upstream tributary Dry Creek (IDNR, 2013). Both reaches had reported aquatic life and primary contact impairments due to biological and indicator bacteria stressors that indicate average to below-average water quality (IDNR, 2013). We estimated the sensitivity of WTP for changes in P to variation in baseline water quality. We took the national mean water quality ladder value reported in Johnston and Besedin (2009) and then parameterized the model using one standard deviation below (2.13 on the RFF ladder) and above (4.60 on the RFF ladder) the meta-analysis' mean value, to derive a lower and higher bound WTP. We applied these low and high benefits transfer models to a scenario of a 50% reduction in P loading, which corresponds to a 2-step improvement on the RFF water quality ladder. The function yielded estimates of $44.56 and $60.52 WTP values per household for 50% reductions in P, for the low- and high-bound cases respectively.

To apply these WTP values to the modeled P changes in each of our scenarios, we prorated each WTP value assuming a nonlinear relationship between it and the modeled changes in P loading based on a power function of the form $f(x)=WTP_{low}=\alpha^{*}x^{1/2}$, where the nonlinear scaling coefficient α (63.0 and 85.6 for the low and high bound models respectively) was calculated to match the WTP for a 50% reduction from the benefit function, as outlined by Johnson et al. (2012) and applied again in Meehan et al. (2013). Adjusting the low and high values in this way assumed WTP per household decreases as reductions in P loading increase. We found that increased area of CRP reduced phosphorus loading significantly, providing benefits with net present values ranging from $16.7 M to $17.3 M.

2.4.3. Nitrate and drinking water quality

The methods we used for phosphorus and sediment valuation were specific to changes in surface water quality. However, changes in land use and management can also impact groundwater quality. When nitrogen-based fertilizer is applied to cropland, it is readily converted to nitrate, a species of nitrogen (N) that is highly mobile and easily leached from soils into groundwater. Natural levels of nitrate in groundwater are low and elevated N in groundwater is almost always associated with inputs from human activities (Spalding and Exner, 1993; Tomer and Burkart, 2003). Nitrate that leaches into groundwater is a public health concern when it contaminates groundwater aquifers that serve domestic households or public water suppliers. The U.S. Environmental Protection Agency (EPA) set the maximum contaminant level (MCL) for nitrate at 10 ppm nitrate-N under the Safe Drinking Water Act (SDWA) due to links between consumption of high levels of nitrate and methemoglobinemia (blue baby syndrome) and some forms of cancer (Weyer et al., 2008; Brender et al., 2013).

All residents within the Indian Creek watershed rely on groundwater for their drinking water source, so changes in the quality of tapped aquifers and elevated nitrate levels can directly affect households living in the watershed (Maupin et al., 2014). We obtained county-level estimates of the total population served by public supply and domestic self-supplied groundwater. About 90% of Indian Creek households are served by public water suppliers and 10% are supplied by domestic wells (Maupin et al., 2014). We applied separate valuation approaches to estimate potential costs to private domestic well owners versus households served by public water suppliers.

Although individual households with private groundwater wells are not required to comply with EPA standards, we assume the majority of residents value drinking water with safe levels of nitrate. For these private wells, we assessed the value of changes in groundwater nitrate concentrations that are predicted to result from changes in nitrogen loading associated with the extent of agricultural land use and CRP. If a domestic well is contaminated with nitrate, residents can avoid drinking contaminated water by digging a new deeper well, by purchasing bottled water, or by installing a nitrate removal system to serve their home water supply. Each of these alternatives comes with costs, including upfront costs from installation or construction and annual operations and maintenance costs.

To estimate baseline groundwater contamination for private wells in the Indian Creek Watershed, we obtained domestic well data with coordinate locations and nitrate records since 1984 from the Linn County, IA GIS office. This dataset includes 1150 unique well records, or approximately 40% of private drinking wells, located within the Indian Creek watershed (Maupin et al., 2014). These data indicate that 1.7% of wells report maximum nitrate records above 10 ppm nitrate-N. We extrapolated this same level of contamination to other private wells in Indian Creek for which records do not exist and assessed the cost of mitigating this level of contamination on all households in Indian Creek that rely on self-served groundwater. We assumed that the percent of known contaminated wells (1.7%) reflects conditions under the baseline scenario and then estimated well contamination and costs for each alternative scenario by linearly scaling costs as a function of modeled changes in surface N loading. For example, for a scenario where N export increased by 1.6% (as is the case in the ‘Partial CRP’ scenario), we applied a corresponding 1.6% change in the frequency of contamination to translate modeled changes in N export from surface water using SWAT to changes in N contamination in groundwater.

We evaluated the costs of well nitrate contamination using both contingent valuation and avoided costs approaches. First, we applied contingent valuation survey results from Crutchfield et al. (1997) that found the average US household is willing to pay $1036 per year to avoid exposure to elevated nitrate in their drinking water. We applied this average per household WTP to the number of households predicted to be exposed to groundwater with nitrate concentrations exceeding 10 ppm nitrate-N under each alternative scenario. This approach provided an upper bound estimate of the value of avoided contamination of drinking water wells.

Second, we applied an avoided cost approach to estimate household costs associated with actions to reduce or avoid nitrate contamination. The benefits of reduced groundwater nitrate contamination include avoided costs of remediation actions taken to replace a contaminated well, install a filtration system or other treatment technologies, or purchase bottled water for drinking and cooking (Lewandowski et al., 2008). In a Minnesota survey, Lewandowski et al. (2008) found that roughly equal proportions of households with well nitrate contamination choose to treat water, dig a new deeper well, or purchase bottled water. The survey was conducted in conjunction with well nitrate testing so survey responses were compared to actual actions taken by owners with confirmed cases of nitrate contamination. For our analysis, we assumed that affected households in Indian Creek would treat for nitrate following the observed adoption rates reported in Lewandowski et al. (2008) and applied those estimated costs to the number of predicted contaminated wells under each scenario. The estimated average cost was $6261 per household over a ten-year time horizon with a 3% discount rate. This approach provided a lower bound estimate of the value of avoided nitrate contamination of private wells in Indian Creek. In this analysis we found that reduced nitrate loading provided private well drinking water benefits between $36,644 to $93,667.

Public water suppliers, defined as those systems that provide water for human consumption with at least 15 service connections or serve at least 25 people at least 60 days during the year, are subject to comply with the nitrate MCL, enforced at the state level by the Iowa Department of Natural Resources (IDNR). These systems may face reporting and health-based violations if they either fail to monitor for nitrate in their source water wells as required or exceed the MCL in the finished water supplied to customers. Compliance to the EPA standard means that suppliers must proactively reduce risks from nitrate contamination and may face costs associated with drilling new source water wells or installing ion exchange or reverse osmosis treatment systems, to reduce their risk of violating the SDWA (Honeycutt et al., 2012; Jensen et al., 2012).

Of the fifteen active public water suppliers in Indian Creek, data from the Iowa Natural Resources Geographic Information System (NRGIS) Library indicate that thirteen systems operate their own source water wells and two purchase water from other nearby communities (IDNR, 2015). For the subset of suppliers which operate their own source water wells, we evaluated the best-available data on baseline nitrate levels from two sources: (1) annual mean concentrations of finished water from community systems that are reported nationally to the Centers for Disease Control and Prevention (CDC) Environmental Health Tracking Program (CDC, 2014) and (2) maximum nitrate concentrations from sampled public water supply source water wells reported in a state-level well sampling record database, which only includes data on a portion of the watershed's existing source water wells (IDNR, 2015). Using the maximum concentration of N observed in either dataset, we found that none of the available data demonstrated levels of nitrate for public water suppliers in the watershed currently above the MCL (e.g. the maximum for any system was a nitrate sample of 5.87 mg/L from a well operated by the City of Hiawatha). None of these systems are actively treating or are required to treat for nitrate currently (verified through personal correspondence with IDNR February 2015), though the majority of the area's public water suppliers do appear to monitor their finished and source water nitrate levels. To estimate adjusted nitrate concentrations for each of these public water suppliers across the different CRP scenarios, we calculated estimated maximum nitrate concentrations by assuming the percent change in modeled surface water N loads corresponded linearly to the maximum nitrate concentrations for each public water supplier.

While, as noted above, no public water supply system within the Indian Creek study area currently treats for nitrate, the vulnerability of public water supply wells to contamination may change current conditions under scenarios of expanded or reduced CRP, as a result of expected changes in cropland and associated fertilizer applications. To analyze this relationship, we estimated the percent change in cropland area from the current CRP scenario for those public water supply wells in the study area for which delineated groundwater capture zones were available (IDNR, 2015). Since capture zones with thick confining layers are not vulnerable to changes in nitrate export, we considered suppliers to be susceptible if their capture zones have confining layers less than or equal to 50 feet above the aquifer, supported by IDNR's definition of aquifer susceptibility (IDNR, 2015). By this definition of vulnerability, we found that seven of the fifteen public water suppliers in Indian Creek have vulnerable groundwater supplies. We then assumed that those suppliers experiencing increased cropland within their groundwater capture zones in the ‘Reduced CRP’ and ‘No CRP’ scenarios compared to baseline land cover to be at greater risk for groundwater nitrate contamination, and that these suppliers could face additional costs to protect or treat their source water as a result. We assumed suppliers for which cropland decreased within the capture zones of their vulnerable wells under the expanded CRP scenarios could avoid costs due to a potentially lower risk from groundwater contamination.

Applying this risk-mitigation approach, we identified two out of the seven vulnerable public water suppliers that could be at higher risk to N loading as a result of increased cropland around their groundwater capture zones under reduced CRP scenarios: Marion Municipal Water Department, the largest public water supplier in Indian Creek that provides drinking water to almost 35,000 people residing in the City of Marion and other neighboring communities, and Meadow Knolls Addition, a smaller supplier in the watershed serving less than 100 people. If both of these suppliers installed ion exchange facilities (systems which tend to be more cost effective for lower baseline nitrate levels and smaller suppliers than reverse osmosis treatment, another common alternative) to reduce this risk to their source water, we estimated that, in 2013 dollars, the Marion Water Department could incur costs equivalent to a net present value between $2.3 million and $18.5 million, and Meadow Knolls could incur costs between $80,000 and $1.3 million, over ten years (using a 3% discount rate). These cost ranges were calculated using national, low- and high-

Table 1
Valuation of benefits provided by CRP scenarios compared with baseline Current CRP. 'Water Quality' incorporates the sum of three types of water quality benefits calculated using different approaches; 'Air Quality' presents the sum of public health impacts of both NOx and NH_3; 'Greenhouse gas reduction' includes the climate change mitigation benefits of both N_2O and CO_2. Net present value calculated in 2013$ for 10-year CRP contract duration and 3% rate of discount.

Scenario	Water quality (P, N, sediment)		Flood damage [a]	Air quality	Greenhouse gas emissions	
	Lower Bound	Higher Bound			Lower bound	Higher bound
No CRP	−2,291,957	−3,099,752	−134,308	−105,542	−3,469,908	−9,464,618
Partial CRP	−2,211,982	−2,992,220	−113,273	−98,049	−2,731,178	−7,449,811
Baseline CRP	0	0	0	0	0	0
Riparian CRP	12,182,002	16,503,228	199,155	107,737	800,293	2,183,877
Wetland CRP	6,892,904	9,316,850	73,667	95,078	957,763	2,613,192
Wetland-Riparian CRP	12,983,363	17,559,669	255,974	189,381	1,058,588	2,889,271

[a] NOTE: Flood Damage calculations were based on a model with 100 m grid resolution so although landuse scenarios were developed at 30 m resolution, 100 m practices were modeled.

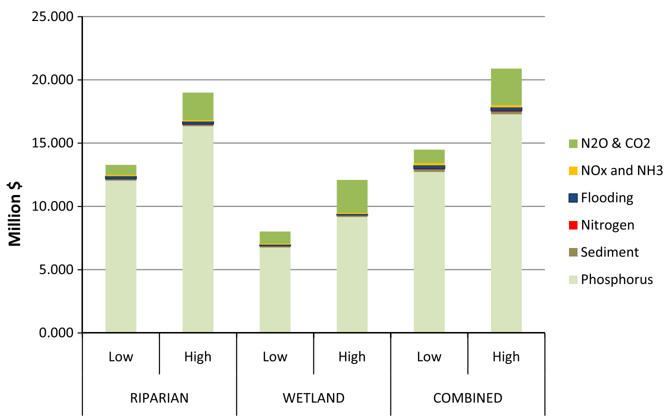

Fig. 2. *Total ecosystem service value.* Total values, using both lower and higher values for ecosystem service valuation, of all analyzed ecosystem service benefits provided by scenarios of increased CRP.

bound estimates of annualized capital and operation and maintenance costs (Jensen et al., 2012) associated with ion exchange treatment facilities appropriate for the size and flow characteristics for both water supply systems. Given insufficient information about how public water suppliers in the study area have responded to increases or decreases in their risk of nitrate contamination in the past, we made the conservative assumption to not include expected or avoided costs to public water suppliers in the final CRP valuation estimates (Table 1 and Fig. 2). Cost estimates are included here to illustrate potential future costs to communities if degradation of water quality in the region continues.

2.4.4. Greenhouse gas emissions and climate change mitigation

We also estimated the potential climate change mitigation benefits provided by CRP. Cultivation of annual crops alters carbon cycling and enhances emissions of potent greenhouse gases (Robertson et al., 2000). Conversion of annual row crops to perennial vegetation increases sequestration of carbon in soils and plant biomass (Johnson et al., 2007). We estimated the potential changes in carbon stored by varying amounts of perennial vegetation associated with increased or decreased acres enrolled in CRP. We used region-specific estimates of average annual rates of carbon sequestration associated with conversion of crops to forests, grasses and wetlands and estimated losses from conversion of perennial vegetation to annual crops (Anderson et al., 2008). Using the land use-land cover data developed by Smith et al. (2013) we calculated the mean annual tons of carbon sequestered by restoration to perennial vegetation associated with the three scenarios of CRP gain, and calculated the one-time carbon loss that would be associated with removal of lands from CRP in the two loss scenarios (Table S4).

To assess the value of changes in carbon storage we applied two values from the US federal government standards for the social cost of carbon (SCC). The SCC accounts for the potential global-scale damages due to climate change attributable to a unit increase in carbon or carbon-equivalent emissions (Tol, 2008). We used the median government standard for the SCC of \$37/Mg CO_2 emitted under a 3% discount rate in 2013, the discount rate recommended by guidance form the Office of Management and Budget (Interagency Working Group on Social Cost of Carbon, 2013). To estimate the value of carbon captured by increased CRP acres we multiplied the estimated annual Mg of CO_2 sequestered by the SCC to determine an annual value, and then calculated a NPV for 10 years' worth of carbon stored during a standard CRP contract. For the two scenarios of reduced CRP we multiplied the instantaneous reduction in stored carbon by the SCC to determine the value of this one-time change in carbon storage. To include an upper bound estimate of the potential climate change mitigation benefits provided by CRP we also used the 95th percentile of SCC estimates from the US government equivalent to approximately \$100/Mg CO_2 (Table 1). Across all increased CRP scenarios and using both upper and lower benefit estimates, the value of changes in carbon sequestration ranged from \$710,317 to nearly \$2.5 M.

In addition to increasing carbon sequestration, conversion of cultivated cropland to perennial vegetation also reduces emissions of the potent greenhouse gas N_2O. After fertilizer is applied to farm fields, some of the applied N is released through soil microbial processes resulting in atmospheric emissions of N_2O (Davidson et al., 2015). Retirement of agricultural land stops fertilizer application and reduces emissions of nitrous oxide (N_2O) contributing to mitigation of global climate change. Using application rates consistent with production practices in Iowa, we assumed that all corn crops are fertilized with an average annual application of 161 kg Ha^{-1} of anhydrous ammonia and calculated annual N_2O emissions based on constant emissions factors specific to anhydrous ammonia (Table S4) (De Klein et al., 2006; Stehfest and Bouwman, 2006). We adjusted total anhydrous ammonia inputs within the watershed as the total area under corn cultivation changed for each CRP scenario (Table S4).

We used the same lower and upper bound SCC values to assess the climate change mitigation benefit of reduced N_2O emissions however we adjusted the value to account for the greater global warming potential of N_2O. Marten and Newbold (2012) suggest that due to differences in long-term radiative forcing, the social cost of N_2O (SCN_2O) is 390 times that of CO_2. Consequently, we estimate the SCN_2O as \$14,414/Mg N_2O and apply a 3% discount rate over 10 years to calculate the NPV of changes in N_2O emissions associated with changing CRP (Marten and Newbold, 2012). We estimated the NPV of the climate change mitigation benefit of reduced N_2O associated with additional CRP acres to range from

$79,403 to $433,555.

2.4.5. NO_x and NH_3 and air quality

Additional societal costs are incurred as a result of emissions from the use of nitrogen-based fertilizers that impact air quality. In addition to N_2O, fertilizer application also emits NO_x, and NH_3, air pollutants that contribute to the formation of N_2O particulate matter ($PM_{2.5}$) in the atmosphere (Davidson et al., 2015). Overexposure to $PM_{2.5}$ can cause damage to the respiratory system and is associated with increased incidence of asthma and premature deaths (Marten and Newbold, 2012; Compton et al., 2011). Nitrogen oxides can also produce ozone (O_3), which can lead to and exacerbate respiratory disease and asthma (Townsend et al., 2003). In contrast to consequences due to climate change, damages to human health from NO_x and NH_3 depend on weather patterns and are generally localized to the geographic source of emission (Birch et al., 2011). We used the same method described above for N_2O to estimate emissions of NO_x, and NH_3 from nitrogen-based fertilizer application for each CRP scenario (Table S4).

We evaluated the benefits of reduced NH_3 emissions from fertilizer application based on the contribution to premature deaths caused by formation and exposure to $PM_{2.5}$ (Krewski et al., 2009). Benefits of reduced NO_x emissions were estimated based on impacts to human health and crop losses due to ozone (O_3) and small particulate matter ($PM_{2.5}$) formation (Delucchi, 2000). The cost of premature death reflects the WTP of people in the US for small reductions in their risk of mortality. According to the USEPA (2013), the "value of a statistical life" (VSL) is $8.55 M assuming 2013 as the currency year and baseline income. To estimate the total cost of damages associated with air pollution emissions, we used the median cost per unit estimates and multiplied by the amount of NH_3 and NO_x emissions for each scenario. The estimated costs of NH_3 and NO_x in 2013$/Kg are $3.28 and $15.80, respectively (Kusiima and Powers, 2010; Delucchi, 2000; Muller and Mendelsohn, 2007). In sum, we estimated NH_3 and NO_x reductions associated with increased CRP would provide air quality benefits worth $95,078 to $189,381.

3. Results and discussion

CRP lands in the Indian Creek watershed provide benefits that exceed the costs of payments to farmers for all scenarios evaluated, even when lower bound values were used to assess the value of some ecosystem service benefits and when ecosystem service values were reduced by 20% to account for the impact of potential leakage associated with increased CRP enrollment. For the scenarios of CRP expansion, the NPV of estimated increases in the value of ecosystem service benefits ranged from $6.4 M for 3752 acres (1,518 ha) of CRP in the Wetland scenario using lower bounds for ecosystem service values, to $16.7 M for 3922 acres (1587 ha) of CRP (in the combined Wetland/Riparian scenario applying higher bounds for ecosystem service values) (Table 2). Loss of the baseline 966 CRP acres (391 ha) would generate losses in ecosystem service values ranging from $5.1 to $10.7 M for the Partial Loss scenario with only 70 CRP acres (28 ha), and from $6 to $12.8 M for the Total Loss scenario with no CRP acres. The value of changes in water quality accounted for the majority of the value of ecosystem services associated with scenarios of increases in CRP; between 77% and 92% of the total value (depending on the scenario and valuation methods) was associated with water quality benefits resulting from reduced P, N and sediment loading (Fig. 2). However, the value of greenhouse gas emissions accounted for between 53% and 74% of the total change in ecosystem service value associated with the two scenarios of CRP loss due to the significant up-front carbon losses associated with conversion of perennials to row crops. The most significant component of the water quality value relates to the reduction in P loading and the associated increase in benefits related to improved surface water quality (Table 1). These large values for changes in surface water quality result primarily from the significant reductions in P loading modeled by SWAT; the Riparian and Combined Wetland/Riparian scenarios achieved 55% and 61% reductions in P from the baseline scenario respectively (Table S3). As a result of these large modeled changes in loading, we estimated increased water quality benefits from P reduction ranging from an NPV of $6.7 M using a lower bound for the Wetland scenario to an NPV of more than $17 M using a higher bound for the Combined Wetland/Riparian scenario. Removing CRP entirely from Indian Creek would correspond with a loss in P-related water quality benefits ranging from $2.2 M to $3 M.

Although the additional CRP lands modeled in the Combined Wetland/Riparian scenario provided the greatest absolute increase in ecosystem service benefits, other scenarios proved more cost-efficient and provided greater net benefits. In particular, the Riparian scenario, which added 1408 acres (570 ha) of grassland buffers to CRP, generated benefits (accounting for 20% leakage) ranging from $4478 per acre ($11,061 per ha) using lower bound values to nearly $6401 per acre ($15,810 per ha) using higher bound values. The Combined Wetland/Riparian scenario also provided benefits that greatly exceeded the costs of CRP enrollment by increasing grassland CRP by 230 acres (93 ha) and wetland CRP by 2626 acres (1063 ha). Although this combination of restored perennial vegetation still provided a good return on investment, this scenario was less cost-effective and generated benefits between $2955 and $4262 per acre ($7,299 and $10,527 per ha).

Our analysis of the multiple benefits provided by retired agricultural lands in Indian Creek demonstrates that CRP in this watershed provides a positive return on investment. Improvements in water quality, in particular associated with reduced P loading, provide the greatest source of estimated benefit from increased CRP, and targeting investment in riparian areas would be the most cost-effective strategy for attaining these benefits. These findings

Table 2
Comparison of NPV of benefits and costs of CRP across alternative scenarios, 2013$.

Scenario	Change in ecosystem service benefits assuming 20% leakage		Change in CRP rental costs	**Benefits – costs of decreasing/increasing CRP from Baseline**	
	Lower bound	Higher bound		**Lower bound**	**Higher bound**
No CRP	−6,001,715	−12,804,220	−974,849	**−3,826,523**	**−9,268,527**
Partial CRP	−5,154,482	−10,653,353	−909,371	**−3,214,214**	**−7,613,311**
Baseline CRP	0	0	0	**0**	**0**
Riparian CRP	10,631,349	15,195,197	1,879,383	**8,751,966**	**13,315,814**
Wetland CRP	6,415,529	9,679,029	4,183,988	**2,231,542**	**5,495,042**
Wetland-Riparian CRP	11,589,845	16,715,436	4,344,833	**7,245,012**	**12,370,603**

are consistent with previous research demonstrating multiple environmental benefits of CRP (Drum et al., 2015; Sullivan et al., 2004; Gleason et al., 2011; Rao and Yang, 2010; Gelfand et al., 2011).

Importantly, we found positive returns on investment in CRP even though we were not able to quantify all of the ecosystem service benefits of this program. For example, we did not estimate the value of water supply for domestic and manufacturing use, impacts to gulf hypoxia, degraded aesthetic quality, or non-use values (Loomis et al., 2000). We also did not include the estimated value of avoiding potential future treatment costs incurred by public water suppliers attributable to CRP adoption and protected water quality. Valuation of habitat-related services could reveal additional benefits provided by CRP, and depending upon the habitat type and species assessed, such analysis could attribute greater values to wetland restoration and potentially shift the ranking of scenarios. Additionally, CRP provides on-site soil retention benefits that lead to increased yields and reduced use of inputs for farmers (Sullivan et al., 2004). Although we valued the public benefits of surface water quality improvements resulting from reduced sediment loading, we did not incorporate the private value associated with increased productivity, and such yield gains are significant enough to often prompt higher land values for farmland with CRP enrollment (Wu and Lin, 2010).

There are several important assumptions inherent in this analysis. First, there is still considerable uncertainty in the economic valuation of ecosystem services (Johnson et al., 2012). For example, although stated preference methods are widely used and improving, there remain criticisms of this approach and its reliability (Wegner and Pascual, 2011). As the stated preferences for water quality improvements constitute the largest portion of estimated ecosystem service values in our analysis, variations in this value would have significant implications for the overall assessment of the benefit-cost ratio of CRP. Additionally, changes in our assumptions about the linear relationship between the biophysical changes in nutrient and sediment loading and the numbers of households potentially affected could significantly change the estimated values of water quality improvements achieved by the CRP. The short time interval of this study of ten years (which is a standard CRP contract length) also assumes no delay between changes in vegetation and agricultural practices and changes in water quality constituents and impacts on beneficiaries. Although this is not the case, we think the conclusion nonetheless holds that investment in land retirement is justified because over a longer time interval biophysical changes will eventually influence changes in ecosystem services and benefits.

Notwithstanding uncertainty associated with valuation, as well as with biophysical models and the linkages between biophysical changes and human well-being, this analysis suggests that at the current level of rental payment and accounting for 20% leakage, increased investment in CRP in Indian Creek would provide benefits that exceed the costs of payments to farmers, by a factor of between 1.5 and 8.1. Further divestment in this program would produce a net cost to society by eroding the ecosystem services provided by CRP. Benefits provided by CRP are significant enough that a higher acreage cap and higher rental payments may be justified and necessary to ensure continued enrollment in the context of higher commodity prices. The assessed scenarios were not equally cost-effective and our analysis shows that increased investment in riparian buffers may be particularly justified. Additional analysis of the biophysical performance and economic value to beneficiaries could inform targeting of lands that are particularly valuable in terms of provision of public ecosystem service benefits and ensure that investment in CRP in Indian Creek and elsewhere continues to provide a positive return on taxpayer investment.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at http://doi:10.1016/j.ecoser.2016.03.004.

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