Agricultural Water Management 96 (2009) 925–930

Contents lists available at ScienceDirect

Agricultural Water Management

journal homepage: www.elsevier.com/locate/agwat

Annual carbon and nitrogen loadings for a furrow-irrigated field

A.P. King [a], K.J. Evatt [a], J. Six [b], R.M. Poch [c], D.E. Rolston [a], J.W. Hopmans [a,*]

[a] Department of Land, Air and Water Resources, University of California Davis, One Shields Ave., Davis, CA 95616 USA
[b] Department of Plant Sciences, University of California Davis, One Shields Ave., Davis, CA 95616 USA
[c] Departament de Medi Ambient i Ciències del Sòl, Universitat de Lleida, Catalonia, Spain

ARTICLE INFO

Article history:
Received 12 August 2008
Accepted 6 January 2009
Available online 8 February 2009

Keywords:
Runoff
Dissolved organic carbon
Carbon sequestration
Reduced tillage
Sediment transport

ABSTRACT

Evaluations of agricultural management practices for soil C sequestration have largely focused on practices, such as reduced tillage or compost/manure applications, that minimize soil respiration and/or maximize C input, thereby enhancing soil C stabilization. Other management practices that impact carbon cycling in agricultural systems, such as irrigation, are much less understood. As part of a larger C sequestration project that focused on potential of C sequestration for standard and minimum tillage systems of irrigated crops, the effects of furrow irrigation on the field C and N loading were evaluated. Experiments were conducted on a laser-leveled 30 ha grower's field in the Sacramento valley near Winters, CA. For the 2005 calendar year, water inflow and runoff was measured for all rainfall and irrigation events. Samples were analyzed for C and N associated with both sediment and dissolved fractions. Total C and N loads in the sediment were always higher in the incoming irrigation water than field runoff. Winter storms moved little sediment, but removed substantial amounts of dissolved organic carbon (DOC), or about one-third of the total C balance. Despite high DOC loads in runoff, the large volumes of applied irrigation water with sediment and DOC resulted in a net increase in total C for most irrigation events. The combined net C input and N loss to the field, as computed from the field water balance, was 30.8 kg C ha^{-1} yr^{-1} and 5.4 kg N ha^{-1} yr^{-1} for the 2005 calendar year. It is concluded that transport of C and N by irrigation and runoff water should be considered when estimating the annual C field balance and sequestration potential of irrigated agro-ecosystems.

1. Introduction

Soils are the largest C pool in the terrestrial environment, containing two to three times the carbon that is associated with the aboveground plant biomass (Entry et al., 2002). Soil management as a means to sequester C has been widely studied to reduce CO_2 fluxes to the atmosphere with the goal to mitigate global warming effects. However, most of these soil C studies were undertaken to evaluate practices that minimize soil respiration, such as minimum tillage. Much less information is available on the impacts of irrigation on the carbon budget in agricultural systems.

The contribution of irrigation water to field C and N dynamics has been quantified for few agricultural systems (Leuking and Schepers, 1985; Dersch and Bohm, 2001; Entry et al., 2002; Gillabel et al., 2007). Data are even more limited for Mediterranean climates such as in California, where irrigation is absolutely required for summer crop production. The role of irrigation in C sequestration is largely undecided. Despite the historical lack of agreement, there is growing consensus worldwide that irrigation will increasingly sequester soil C at the field scale (Follett, 2001; Lal, 2001; Dormaar and Carefoot, 1998; Coneth et al., 1998; Gillabel et al., 2007; Lal et al., 1998). Irrigation promotes biomass accumulation in crops, producing additional C in the crop residues to stabilize soil C. Concurrently, the added water enhances microbial activity, increasing mineralization rates and subsequent CO_2 releases to the atmosphere. These opposing biological processes can confound estimates of the effect of irrigation on soil C storage. Entry et al. (2002) estimated that if the area of irrigated agricultural land was increased by 10% globally, and in addition the equivalent amount of land was returned to native grassland, 3.4×10^9 Mg C could be sequestered, equivalent to approximately one-sixth of global C emissions over the next 30 years.

Long-term experiments in Austria have shown an additional loss of 2.4 ton C ha^{-1} over 21 years in irrigated systems compared to rain-fed treatments (Dersch and Bohm, 2001) due to increased mineralization during otherwise dry periods. In addition, there can be extensive energy costs associated with pumping and delivering irrigation water. The C emissions from these operations are generally not accounted for in analyses of the contribution of

* Corresponding author. Tel.: +1 530 752 3060; fax: +1 530 752 5262.
E-mail addresses: apking@ucdavis.edu (A.P. King), kjevatt@ucdavis.edu (K.J. Evatt), jwsix@ucdavis.edu (J. Six), rosa.poch@macs.udl.cat (R.M. Poch), derolston@ucdavis.edu (D.E. Rolston), jwhopmans@ucdavis.edu (J.W. Hopmans).

0378-3774/$ – see front matter © 2009 Elsevier B.V. All rights reserved.
doi:10.1016/j.agwat.2009.01.001

irrigation to C sequestration (Schlesinger, 1999). Carbon dioxide is also released by degassing of bicarbonated waters from aquifers when they are used for irrigation (Schlesinger, 1999, 2000). Therefore, regardless of tillage type or intensity, irrigation is generally not considered as a soil C sequestration option at the regional planning scale, since the CO_2 emissions can be greater than the equivalent increase in soil organic carbon (SOC) (ECCP, 2003).

Very little information is available on C and N transport in irrigated fields, which is the focus of the current study. Christopher and Lal (2007) and Izaurralde et al. (2007) noted that eroded C could be of the same order of magnitude as other ecosystem processes controlling C flux, such as tillage and microbial decomposition. Erosion may also lead to the stabilization of C in depositional sites (Van Oost et al., 2007). If irrigation water contains dissolved organic carbon (DOC) and/or sediment-associated C, the increased soil organic matter (SOM) input can be a direct source of soil C. This C may indirectly enhance C sequestration by promoting aggregate formation (De Gryze et al., 2005; Kong et al., 2005), thereby providing for the physical protection of OM inside these newly formed aggregates. For example, Gillabel et al. (2007) showed that increased SOM in irrigated systems was due strictly to an increase in microaggregate-associated C.

In California's Sacramento Valley, row crops are generally furrow-irrigated with surface water. At the farm level, water is delivered to fields through a system of natural sloughs and open ditches. Runoff is directed back into the system of sloughs, eventually draining into the Sacramento River delta. Efficient management of nutrient inputs and irrigation water is thus of critical importance to the delta ecosystem. Constituents such as total suspended solids (TSS), NO_3-N, NH_4-N, and DOC are of concern for a variety of reasons pertinent to human and wildlife health. TSS in streams is the primary pollutant of concern for anadromous fish species attempting to spawn in river beds (Cordone and Kelley, 1961; Kondolf, 2000). NO_3 and NH_4 above established maximum levels for drinking water quality standards are toxic to humans. DOC is of particular concern for the urban population of agricultural areas, as it reacts with chlorine in the water treatment process to form trihalomethanes, which are known carcinogens (Fujii et al., 1998). All of these constituents in agricultural runoff are of concern to growers, who must manage irrigated fields to reduce the impact of agriculture on regional water quality. Surface water runoff is typically generated in the winter months (December–March), when most of the annual precipitation falls, and throughout the irrigation season (April–September).

This study was part of a field scale C sequestration project that focused on C sequestration potential for standard and minimum tillage systems of irrigated crops. Greenhouse gases, crop and weed biomass production and soil were sampled in concert throughout the calendar year to determine the C and N budget for an irrigated field. The present study follows a preliminary evaluation of the effect of furrow irrigation on the C and N budget of the same irrigation system (Poch et al., 2006), but limited to the two final irrigations of the 2004 growing season. The current study evaluates water analyses for the 2005 calendar year. The primary objectives were (1) to determine whether the short duration results of Poch et al. (2006) were repeatable the following year, and (2) to quantify the contribution of irrigation and storm water to C and N loads of the irrigated field.

2. Materials and methods

The study was conducted on a laser-leveled 30 ha grower's field (Field 74) in the Sacramento Valley near Winters, CA. Two soil types are present on the field: 70% of the field is Myers clay (fine, montmorillonitic, thermic, entic Chromoxerert) and 30% is Hillgate loam (fine, montmorillonitic, thermic, typic Palexeralf). Details of the soils can be found in Poch et al. (2006). Field 74 was the site of a multi-disciplinary experiment designed to evaluate the effect of tillage treatment on greenhouse gas emissions and C sequestration potential. In the fall of 2003, the field was split in half, using conventional and no-till treatments in the northern and southern half of the field, respectively. Following the harvest of a maize crop in September 2004, the field was left fallow for the winter. In May 2005, the field was planted with sunflower (*Helianthus annus* L.). This research reports on the 2005 calendar year, for which tillage, irrigation and pest management operations are summarized in Table 1. Poch et al. (2006) determined no differences in water quality parameters between the standard and minimum tillage treatments. In addition, differences between tillage treatments were greatly reduced in spring 2005 due to grower operations. For these reasons, the current study did not separate runoff from the tillage treatments and instead treated Field 74 as a single field.

The field was leveled to a 1% slope (furrows running west to east) and drained to a single drainage outlet (Fig. 1). All irrigation water was delivered from a surface water reservoir by a series of mostly unlined open ditches, and siphoned to the furrows of the experimental field. Delivery and drainage ditches were cut with a tractor shortly before the first irrigation. Because of the complex system of irrigation supply and drainage ditches, natural creeks and sloughs in the Sacramento Valley, irrigation water for any field flows through a number of upstream fields and supply ditches before delivery to the experimental field. At the end of the 2004 irrigation season, the drainage outlet of Field 74 was instrumented with a fiberglass trapezoidal flume (60° 50.8 mm Plasti-Fab, Tualintin, OR) and a stilling well with a pressure transducer (Global Water Instrumentation Inc., Gold River, CA), logging 30 min runoff volumes.

The volume of incoming irrigation water was estimated at the northwest corner of the field from measurements of irrigation ditch dimensions, ditch water level, and irrigation water velocity at the time of sampling. A HOBO (Onset, Bourne, MA) tipping bucket rain gauge was installed near the flume to measure rainfall. The rainfall (1–6) and irrigation (A–D) events and runoff amounts are summarized in Table 2. Three rainfall events (2, 3 and 5) caused flooding of the field, with the volume of runoff exceeding the capacity of the flume, rendering the flume measurements useless. For those three events, it was instead assumed that all rainfall was lost as runoff.

Table 1
Field operations for 2005.

2005 date	Event
April 29–May 2	1 stubble chop pass; seed lines ripped in the N and half of the S
May 6–10	1 stubble chop pass; 3 bed-disk (to 10–15 cm) passes on N, 2 on S
May 14	1 mulcher pass and pre-emergent herbicide (ethalfluralin) incorporated
May 16	Male sunflowers planted
May 18–19	Sprinkler irrigation (25.4 mm)
May 23	Female and additional row of male sunflowers planted
June 5–9	Furrow irrigation (127.8 mm)
June 17–18	Side-dress fertilizer applied (approx. 89.7 kg/ha)
June 18–20	Both fields cultivated for weeds
July 2–6	Furrow irrigation (183.9 mm)
July 23–28	Furrow irrigation (147.6 mm)
July 30	Esfenvalerate insecticide aerial spray
August 12–17	Furrow irrigation (191.0 mm)
August 29	Male sunflowers disked in
October 8–10	Sunflower harvest

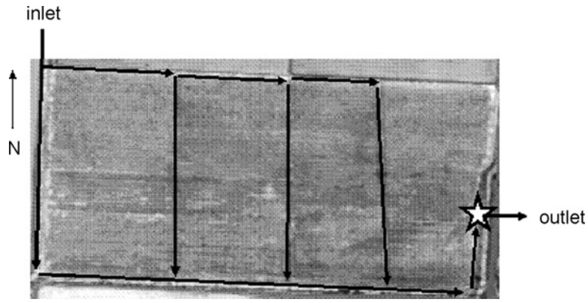

Fig. 1. Diagram of irrigation supply and drainage ditches for experimental field. Field runoff was discharged through the flume at the field outlet.

Three duplicate water samples were collected daily in 1 L bottles from the center of the water column in the delivery ditch at the entrance to the field (incoming) and near the flume at the field drainage site (runoff) for each runoff event (storm or irrigation). Each irrigation event lasted between 4 and 5 days, with farm workers changing irrigated sections of the field beyond our control. Hence, the number of water samples and sampling times were systematically taken at fixed intervals to ensure field representative concentration and sediment load values during the irrigation events. Samples were frozen until analyzed. Defrosted water samples were filtered through a pre-burned 0.3 μm glass fiber filter. Trapped soil material was analyzed for sediment C and N, and water passing through the filter was analyzed for dissolved C and N. Reported constituent concentrations are average values using all samples taken per event, ranging from 3 (for short winter storms that lasted 1 day) to 18 (for irrigation event C, which lasted for 6 days). Standard errors of these averages are reported. Total load values were computed using corresponding discharge measurements; as loads are additive and not averaged, no measurement of error is associated with them.

In addition to inflow and runoff measurements, with corresponding C, N and sediment sampling, Poch et al. (2006) analyzed water samples for $CaCO_3$ in 2004. However, these concentrations were only a negligible fraction of total C. This is not surprising in samples that come entirely from surface water sources where CO_2 levels have equilibrated with that of the atmosphere, and in a region with low carbonate soils. Therefore, $CaCO_3$ was not measured in 2005.

For sediment and filtrate analysis, filters were dried at 60 °C, cooled in a dessicator, and ground sediment was subsequently analyzed for total C and total N on a Carlo-Erba automatic C and N analyzer (Fisons Instrument Model EA/NA 1500 Series 2, San Carlos, CA). Filtrate was divided into two aliquots: one for C and one for N analysis. DOC was measured by a Phoenix 8000 Dohrmann UV Persulfate TOC Analyzer (Mason, OH). Filtrate for N analysis was further divided to analyze for NO_3-N (Doane and Horwath, 2003), NH_4-N (Forster, 1995), and total dissolved nitrogen (TDN) (Cabrera and Beare, 1993). Dissolved organic nitrogen (DON) was calculated by subtracting the inorganic N from the TDN value.

3. Results and discussion

3.1. Sediment

Total C and N loads associated with sediment were always much higher in the irrigation water than in the field runoff (Fig. 2). This same pattern was observed for the limited data of the 2004 maize season (Poch et al., 2006), though increasing erosion and sediment loss early in the season after tillage operations are supposed. Although regional variations between fields and irrigation waters may be expected, this pattern of net sediment gain was consistent across all events sampled at this site. Since C:N ratios were very consistent (Fig. 2a and b), it is believed that the dominant source of C and N was organic material (likely algae) with a relatively constant C:N ratio of 16 $\pm$ 6.

In total, winter storms moved little sediment, thus contributing little to total C and N loads, despite the flooding and high runoff volumes of events 2, 3 and 5 (Table 2). This may be due to the substantial weed cover during the fallow season in the Sacramento Valley's mild climate. While the concentration of total suspended solids (TSS) in the samples varied among runoff events (Table 3), the total loads of suspended solids (SL) were always much higher in the incoming irrigation water. Variation among events was caused by the volume of water applied to the field for each irrigation. As

Table 2
Field water balance of winter storm (1–6) and summer irrigation (A–D) events that caused runoff in 2005. Italicized numbers assume 100% runoff from the field.

Event	Period	Precipitation/irrigation m^3 (cm)	Runoff m^3 (cm)	% lost as runoff
1	29 December–1 January	11571 (3.9)	5280 (1.8)	45.6
2	2 January–4 January	8731 (2.9)	*8731 (2.9)*	*100*
3	10 January–12 January	4007 (1.3)	*4007 (1.3)*	*100*
4	16 February	6000 (2.0)	674 (0.2)	11.2
5	19 February–22 February	7215 (2.4)	*7215 (2.4)*	*100*
6	22 March–23 March	10556 (3.5)	1904 (0.6)	18.0
A	5 June–9 June	37146 (12.4)	14963 (5.0)	40.3
B	1 July–6 July	53462 (17.9)	13652 (4.6)	25.5
C	23 July–29 July	42909 (14.4)	11342 (3.8)	26.4
D	12 August–16 August	55547 (18.6)	8462 (2.8)	15.2

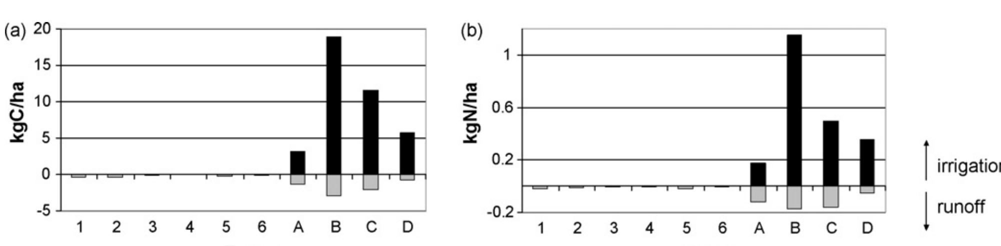

Fig. 2. Total carbon (a) and total nitrogen (b) loads in the sediment (kg/ha) of precipitation runoff (events 1–6) and incoming irrigation water and runoff (events A–D).

Table 3
Total suspended solids (TSS) and sediment loads (SL) for all events. A negative value indicates a net sediment loss from the field. Numbers in parentheses are the standard error of the mean concentration.

	Precipitation/irrigation		Runoff		Sedimentation (kg/ha)
Event	TSS (mg/L)	SL (kg/ha)	TSS (mg/L)	SL (kg/ha)	
1			106 (4.3)	19	−19
2			43 (3.5)	13	−13
3			74 (12.6)	10	−10
4			145 (5.6)	3	−3
5			92 (4.5)	22	−22
6			118 (4.0)	8	−8
A	179 (17.9)	190	192 (62.0)	91	99
B	416 (44.7)	750	214 (29.0)	104	646
C	292 (31.3)	790	443 (70.6)	165	625
D	219 (14.0)	416	178 (35.8)	55	361
				Net sediment deposition:	1656 kg/ha

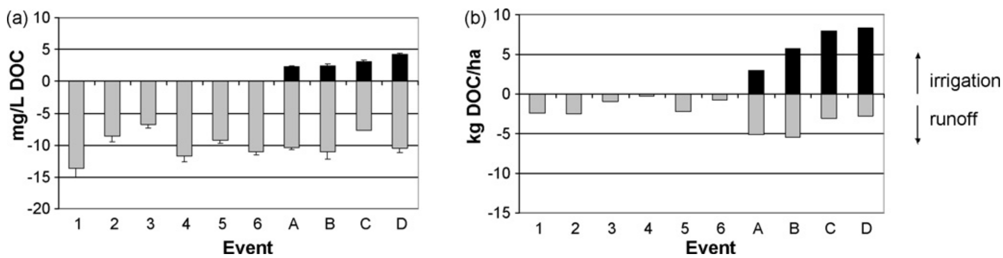

Fig. 3. Dissolved organic carbon concentrations (a) (mg/L) and loads (b) (kg/ha) in the filtrate of precipitation runoff (events 1–6) and incoming irrigation water and runoff (events A–D). Error bars are the standard error of the mean.

was also observed in the 2004 maize season (Poch et al., 2006), a significant sediment mass was retained in the field after each irrigation event, ranging between 99 and 646 kg/ha.

The sediment load of the first irrigation event (A) was much lower than those of the later irrigations. This resulted in a much lower sedimentation rate for event A than the subsequent three irrigations (Table 3). Likely, the incoming water of this first irrigation event was less re-used in upstream agricultural fields than the incoming water of subsequent irrigations. It is possible that the incoming waters later in the irrigation season (events B–D) contained more runoff water from upstream fields, and therefore carried higher sediment concentrations.

3.2. Filtrate

Unlike total C in the sediment, the concentration of DOC in the filtrate was consistently higher in the runoff than the incoming irrigation water (Fig. 3a). A similar result was obtained in the 2004 season study (Poch et al., 2006). DOC is the most mobile source of soil C (Silviera, 2005), and is thus easily affected by land-use change and soil water management. In addition, it has been shown that wet–dry cycles lead to increased soil DOC (Lundquist et al., 1999). California agricultural soils experience severe wet–dry cycles during the hot irrigation season; a crop such as sunflower is only irrigated four times and the surface soil dries out completely in the interim. The winter rainy season is also interrupted by warm dry spells, leading to numerous wet–dry cycles depending on the frequency and duration of rainfall. As the rain/irrigation water passes through the decomposing organic material of the water-carrying furrows, soil material is deposited while the lighter DOC fraction is carried in the water by dissolution from the soil surface (Turchenek and Oades, 1979). This is supported by a highly positive correlation between DOC (kg/ha) of runoff water and runoff volumes ($r = 0.97$, $p < 0.01$). Despite the high DOC concentration in

Table 4
Nitrogen loads in irrigation and runoff water for all events (kg/ha). Dissolved constituents are nitrate-nitrogen (NO_3-N), dissolved organic nitrogen (DON) and total dissolved nitrogen (TDN).

	Precipitation/irrigation				Runoff				Net N movement
Event	NO_3-N	DON	TDN	Sediment N	NO_3-N	DON	TDN	Sediment N	
1					0	0.2	0	0.02	−0.2
2					0.4	0.06	0.4	0.01	−0.9
3					0.04	0.04	0.08	0.01	−0.2
4					0.1	0	0.1	0	−0.3
5					0.02	0.4	0.4	0.02	−0.8
6					0.03	0.1	0.1	0.01	−0.2
A	0.2	2.2	2.5	0.2	0.8	0.8	1.6	0.12	1.7
B	0.4	1.0	1.4	1.2	5.5	1.1	7.1	0.17	−9.9
C	0.8	2.6	3.7	0.5	2.1	0.5	2.7	0.16	2.1
D	0.7	1.8	2.6	0.4	0.8	0.4	1.1	0.05	3.1
								Net N loss from field:	−5.4 kg/ha

Table 5
Carbon loads, as dissolved organic carbon (DOC) and sediment carbon, in irrigation and runoff water for all events (kg/ha).

Event	DOC	Sediment C	Total C
1	−2.4	−0.3	−2.7
2	−2.5	−0.3	−9.3
3	−0.9	−0.1	−2.7
4	−0.3	0	−0.3
5	−2.2	−0.3	−3.5
6	−0.7	−0.1	−0.8
A	−2.2	1.9	−0.3
B	0.3	15.9	16.3
C	4.9	9.5	14.5
D	5.5	5.0	10.5
Net C addition to field:	−0.4	31.2	30.8 kg/ha

storm runoff, the large volumes of applied irrigation water resulted in higher total loads and a net DOC deposition in the field for events B, C and D (Fig. 3b).

The N concentrations in the filtrate samples showed that NH_4^+ levels were insignificant throughout the period of study. The contribution of total dissolved N in the irrigation water (Table 4) was likely a very minor component of the total N field budget. However, following the June 2005 fertilizer application (between events A and B) NO_3-N and DON continued to be present in the runoff water, including in runoff water of the 2006 storm season. The levels of NO_3-N exceeded the drinking water quality standard of 10 ppm (USEPA, 2003) only in event B, immediately after fertilizer application.

3.3. Carbon balance

Table 5 shows the net movement of C in the irrigation and storm waters. As noted earlier, there was a significant loss of DOC from the field during winter storm events. Estimates of DOC losses from non-cultivated temperate ecosystems range from 1 to 146 kg C ha^{-1} yr^{-1} (Hope et al., 1994). Therefore, the measured loss of 9 kg C ha^{-1} yr^{-1} by DOC during the storm season was low, but it is not uncommon (Hope et al., 1994). The majority of C input originated from the sediment of incoming irrigation water. The net C input for the 2004–2005 storm and irrigation season was 30.8 kg C ha^{-1} yr^{-1}, which is higher than that reported for the last 2 irrigation events of 2004 (Poch et al., 2006). The sediment-associated C will likely make a long-term contribution of C input to the field, whereas the infiltrated dissolved fraction is more reactive and mineralizes faster.

Current estimates of C sequestration potential in global agricultural systems vary, and range between 30 and 653 kg C ha^{-1} yr^{-1} (Christopher and Lal, 2007; Hutchinson et al., 2007; Izaurralde et al., 2007; Franzluebbers, 2005; Six et al., 2004; West and Post, 2002). The field-measured estimate of 30.8 kg C ha^{-1} yr^{-1} should thus be seriously considered when calculating the C sequestration potential of irrigated agro-ecosystems. It is realized that this study considers a single field only, and that a rigorous measurement of regional sequestration potential would require measurements of carbon fluxes across multiple fields.

3.4. Nitrogen balance

Nitrogen calculations resulted in a net loss of 5.4 kg N ha^{-1} yr^{-1} from storm and irrigation water movement (Table 4), with the largest losses occurring in irrigation event B, when both NO_3-N and TDN levels were high due to the fertilization prior to irrigation. However, for all other irrigation events, a net N input to the field was measured. Unlike for the C balance, the total dissolved N fraction contributed most significantly to the N balance. Nitrogen loads in the sediment of both irrigation and runoff water were considerably lower than TDN loads, and DON consistently made up the majority of dissolved N constituents in the incoming irrigation waters (Table 4).

4. Conclusions

Irrigation events contributed considerably more sediment and associated C to this irrigated agricultural field than did storm events, largely due to the higher volume of water applied to the field in irrigation events. In total, irrigation and rainfall events accounted for field acquisition of 30.8 kg C ha^{-1} yr^{-1}. While winter storms did not cause enough erosion to mobilize much sediment, storm events did contribute a significant amount of DOC to runoff.

Inorganic fertilizer N did run off the field when irrigation followed a fertilizer application. Timing of fertilizer applications to maximize crop uptake and minimize runoff would reduce runoff concentrations of NO_3-N. Improvement of irrigation efficiency would greatly reduce the runoff of all C and N constituents, and would therefore help minimize non-point source pollution to the Sacramento River delta ecosystem.

In summary, it can be concluded that the contribution of irrigation water to field C and N budgets should be taken into consideration for irrigated agro-ecosystems, particularly for purposes of the evaluation of soil C sequestration potential.

Acknowledgement

This study was part of a larger carbon sequestration study funded by the Kearney Foundation of Soil Science, Project #2005.204.

References

Cabrera, M.L., Beare, M.H., 1993. Alkaline persulfate oxidation for determining total nitrogen in microbial biomass extracts. Soil Science Society of America Journal 57, 1007–1012.
Christopher, S.F., Lal, R., 2007. Nitrogen management affects carbon sequestration in North American cropland soils. Critical Reviews in Plant Sciences 26, 45–64.
Coneth, A., Blair, G.J., Rochester, I.J., 1998. Soil organic carbon fraction in a Vertisol under irrigated cotton production as affected by burning and incorporating cotton stubble. Australian Journal of Soil Research 36, 655–667.
Cordone, A.J., Kelley, D.W., 1961. The influences of inorganic sediment on the aquatic life of streams. Reprint from California Fish and Game, vol. 47, No. 2. California Department of Fish and Game, Inland Fisheries Branch, Sacramento, CA, 41 pp.
De Gryze, S., Six, J., Brits, C., Merckx, R., 2005. A quantification of short-term macroaggregate dynamics: influences of wheat residue input and texture. Soil Biology and Biochemistry 37, 55–66.
Dersch, G., Bohm, K., 2001. Effects of agronomic practices on the soil carbon storage potential in arable farming in Austria. Nutrient Cycling in Agroecosystems 60, 49–55.
Doane, T.A., Horwath, W.R., 2003. Spectrophotometric determination of nitrate with a single reagent. Analytical Letters 36, 2713–2722.
Dormaar, J.F., Carefoot, J.M., 1998. Effect of straw management and nitrogen fertilizer on selected soil properties as potential soil quality indicators of an irrigated dark brown Chernozemic soil. Canadian Journal of Soil Science 78, 511–517.
ECCP (European Climate Change Programme), 2003. Working Group Sinks Related to Agricultural Soils. Final Report. Available at: http://ec.europa.eu/environment/climat/pdf/finalreport_agricsoils.pdf accessed 8/2/2008.
Entry, J.A., Sojka, R.E., Shewmaker, G.E., 2002. Management of irrigated agriculture to increase organic carbon storage in soils. Soil Science Society of America Journal 66, 1957–1964.
Follett, R., 2001. Soil management concepts and soil carbon sequestration in cropland soils. Soil Tillage Research 61, 77–92.
Forster, J.C., 1995. Soil nitrogen. In: Alef, N., Nannipieri, P. (Eds.), Methods in Applied Soil Microbiology and Biochemistry. Academic Press, San Diego, pp. 79–87.
Franzluebbers, A.J., 2005. Soil organic carbon sequestration and agricultural greenhouse gas emissions in the southeastern USA. Soil and Tillage Research 83, 120–147.
Fujii, R., Ranalli, A.J., Aiken, G.R., Bergamaschi, B.A., 1998. Dissolved organic carbon concentrations and compositions, and trihalomethane formation potentials in waters from agricultural peat soils, Sacramento-San Joaquin Delta, California: implications for drinking water quality. U.S. Geological Survey Water-resources Investigations Report 98-4147.
Gillabel, J., Denef, K., Brenner, J., Merckx, R., Paustian, K., 2007. Carbon sequestration and soil aggregation in center-pivot irrigated dryland cultivated farming systems. Soil Science Society of America Journal 71, 1020–1028.

Hope, D., Billet, M.F., Cresser, M.S., 1994. A review of the export of carbon in river water: fluxes and processes. Environmental Pollution 84, 301–324.
Hutchinson, J.J., Campbell, C.A., Desjardins, R.L., 2007. Some perspectives on carbon sequestration potential in agriculture. Agricultural and Forest Meteorology 142, 288–302.
Izaurralde, R.C., Williams, J.R., Post, W.M., Thomson, A.M., McGill, W.B., Owens, L.B., Lal, R., 2007. Long-term modeling of soil C erosion and sequestration at the small watershed scale. Climatic Change 80, 73–90.
Kondolf, G.M., 2000. Assessing salmonid spawning gravel quality. Transactions of the American Fisheries Society 129, 262–281.
Kong, A.Y.Y., Six, J., Bryant, D.C., Denison, R.F., van Kessel, C., 2005. The relationship between carbon input, aggregation, and soil organic carbon stabilization in sustainable cropping systems. Soil Science Society of America Journal 69, 1078–1085.
Lal, R., Kimble, J.M., Follett, R.F., Cole, C.V., 1998. The Potential of U.S. Cropland to Sequester C and Mitigate the Greenhouse Effect. Ann Arbor Press, Chelsea, MI.
Lal, R., 2001. Myths and facts about soils and the greenhouse effect, no.57. Soil Science Society of America Special Publication, pp. 9–26.
Leuking, M.A., Schepers, J.S., 1985. Changes in soil carbon and nitrogen due to irrigation development in Nebraska's sandhill soils. Soil Science Society of America Journal 49, 626–630.
Lundquist, E.J., Jackson, L.E., Scow, K.M., 1999. Wet-dry cycles affect dissolved organic carbon in two California agricultural soils. Soil Biology and Biochemistry 31, 1031–1038.
Poch, R.M., Hopmans, J.W., Six, J.W., Rolston, D.E., MacIntyre, J.L., 2006. Considerations of a field-scale soil carbon budget for furrow irrigation. Agriculture, Ecosystems and Environment 113, 391–398.
Schlesinger, W.H., 1999. Carbon sequestration in soils. Science 284, 2095.
Schlesinger, W.H., 2000. Carbon sequestration in soils: some cautions amidst optimism. Agriculture, Ecosystems and Environment 82, 121–127.
Silviera, M.L.A., 2005. Dissolved organic carbon and bioavailability of N and P as indicators of soil quality. Scientia Agricola 62, 502–508.
Six, J., Ogle, S.M., Breidt, F.J., Conant, R.T., Mosier, A.R., Paustian, K., 2004. The potential to mitigate global warming with no-tillage is only realized when practiced in the long-term. Global Change Biology 10, 155–160.
Turchenek, L.W., Oades, J.M., 1979. Fractionation of organo-mineral complexes by sedimentation and density techniques. Geoderma 21, 311–343.
USEPA (United States Environmental Protection Agency), 2003. National Primary Drinking Water Regulations; List of Drinking Water Contaminants and Their Maximum Contaminant Level. Available at: http://www.epa.gov/safewater/contaminants/index.html#listmcl Accessed November 9, 2008.
Van Oost, K., Quine, T.A., Govers, G., De Gryze, S., Six, J., Harden, J.W., Ritchie, J.C., McCarty, G.W., Heckrath, G., Kosmas, C., Giraldez, J.V., Marques da Silva, J.R., Merckx, R., 2007. The impact of agricultural soil erosion on the global carbon cycle. Science 318, 626–629.
West, T.O., Post, W.M., 2002. Soil organic carbon sequestration rates by tillage and crop rotation: a global data analysis. Soil Science Society of America Journal 66, 1930–1946.