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*Chapter 4*

# MANAGEMENT OF IRRIGATED AGRICULTURE TO INCREASE ORGANIC CARBON STORAGE IN SOILS

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## Abstract

Increasing the amount of C in soils may be one method to reduce the concentration of $CO_2$ in the atmosphere. We measured organic C stored in southern Idaho soils having long term cropping histories that supported native sagebrush vegetation (NSB), irrigated moldboard plowed crops (IMP), irrigated conservation -chisel- tilled crops (ICT) and irrigated pasture systems (IP). The $CO_2$ emitted as a result of fertilizer production, farm operations and $CO_2$ lost via dissolved carbonate in irrigation water, over a 30 year period, was included. Net organic C in ecosystems decreased in the order IP>ICT>NSB>IMP. In this study, if NSB were converted to IMP, 0.15 g C $m^{-2}$ would be emitted to the atmosphere, but if converted to IP 3.56 g C $m^{-2}$ could be sequestered. If IMP land were converted to ICT, 0.95 g C $m^{-2}$ could be sequestered in soil and if converted to IP 3.71 g C $m^{-2}$ could be sequestered. There are $2.6 \times 10^8$ ha of land worldwide presently irrigated. If irrigated agriculture were expanded 10% and the same amount of rainfed land were converted back to native grassland, an increase of $3.4 \times 10^9$ Mg C (5.9% of the total C emitted in the next 30 yr) could potentially be sequestered. The total projected release of $CO_2$ is $5.7 \times 10^{10}$ Mg C worldwide during the next 30 years. Converting rainfed agriculture back to native vegetation while modestly increasing areas in irrigated agriculture could have a significant impact on $CO_2$ atmospheric concentrations while maintaining or increasing food production.

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## Abbreviations

| | |
|---|---|
| C | carbon; |
| NSB | native sagebrush; |
| IMP | irrigated moldboard plowed crop land; |
| ICT | crop land managed with conservation (chisel) tillage; |
| IP | irrigated pasture; |
| $C_S$ | carbon projected to sequestered in soils worldwide over the next 30 years; |
| $C_{EW}$ | carbon projected to be emitted worldwide over the next 30 years. |

## Introduction

In 1992, nearly all countries of the world signed the Framework Convention on Climate change. Its long term goal is to stabilize the concentration of greenhouse gases in the atmosphere at concentrations that should prevent dangerous anthropogenic interference with the climate system. To stabilize or reduce $CO_2$ concentrations, the gas must be transferred from the atmosphere to marine or terrestrial ecosystems. Processes or activities that remove greenhouse gases from the atmosphere are defined as sinks in the Framework Convention. In the 1997 Kyoto Protocol, agricultural soils are specifically recognized in the list of potential sinks of greenhouse gases.

Soils are the largest pool of C in the terrestrial environment (Jobbagy and Jackson, 2000; Schlesinger, 1990; 1995). The amount of C stored in soils is twice the amount of C in the atmosphere and three times the amount of C stored in living plants (Schlesinger, 1990; 1995; Kimble and Stewart, 1995), therefore, a change in the size of the soil C pool could significantly alter the atmospheric $CO_2$ concentration (Wang et al., 1999). The current concentration of C in soils reflects the balance between past C accumulation and loss. Accumulation of C in soils is derived from litter and root input, while losses result from microbial degradation of organic matter, eluviation and erosion (Entry and Emmingham, 1998). At equilibrium, the rate and amount of C added to the soil via vegetation are equal to the rate and amount of C lost through organic matter degradation and other losses (Henderson, 1995). Within limits, soil C increases with increasing soil water and decreasing temperature (Hontoria et al., 1999; Wang et al., 1999; Burke et al., 1989). The effect of soil water is much greater than the effect of soil temperature (Birch and Friend, 1956; Hontoria et al., 1999; Liski et al., 1999). Increasing water within temperature zones can increase plant production and, thus, C input to soils via increased plant litter and root production (Liski et al., 1999).

Land use changes can impact the amount of C stored in the soil by altering C inputs and losses. In forest, grassland and wetland ecosystems, conversion of native vegetation to agricultural cropping has resulted in substantial C transfer to the atmosphere as a result of loss of climax vegetation to the lower equilibrium C concentration in soil (Lal et al., 1999; Wang et al., 1999; Cambardella and Elliot, 1992; Johnson, 1992). In arid and semi-arid environments plant survival and growth is limited by available water and irrigation is required to increase plant production to the point where crops become economically viable. Irrigation also increases C input to soils via increased litter and root production.

When assessing the potential of irrigation of arid or semiarid land to increase C storage in soils, one needs to assess C loss from $CO_2$ emitted to the atmosphere as a result of 1) fertilizer manufacture, storage, transport and application, 2) fossil- fuel $CO_2$ emitted from pumping irrigation water, 3) farm operations, such as tillage and planting, and 4) $CO_2$ lost via dissolved carbonate in irrigation water (West and Marland, 2001; Schlesinger, 1999). Schlesinger (1999) used a fertilization value of 336 kg N $ha^{-1}$ $yr^{-1}$, which is an unusually high fertilization rate in U.S. farms. The $CO_2$ released during fertilizer production of 336 kg N $ha^{-1}$ $yr^{-1}$ is approximately 16.7 g C $m^{-2}$ $yr^{-1}$ (Schlesinger, 1999). It has been noted that a more realistic rate is 100-150 kg N $ha^{-1}$ $yr^{-1}$ (West and Marland, 2001). Carbon dioxide released from pumping irrigation water in the United States ranges from 126 kg C $ha^{-1}$ $yr^{-1}$, when using gasoline, to 266 kg C $ha^{-1}$ $yr^{-1}$ when using electricity (West and Marland, 2001). In addition, C may be lost as $CO_2$ from the irrigation water itself. Irrigation water in arid and semiarid regions often contains as much as 1% dissolved $CO_2$. When water is applied to the soil, $CaCO_3$ can precipitate, releasing $CO_2$ into the atmosphere. If irrigation water containing 0.05 g $L^{-3}$ dissolved Ca is used to irrigate crops in semi-arid climates, the calculated increase in plant C is 2000 g C $m^{-2}$ per year over C contained in native soils and vegetation. The net $CO_2$ released via irrigation water is calculated to be 8.4 g C $m^{-2}$ $yr^{-1}$ (Schlesinger,1999).

Farm management practices, including conservation tillage and erosion control, have reduced the amount of $CO_2$ emitted to the atmosphere in both Canada and the United States (West and Marland, 2001; Janzen et al., 1997; Paustian et al., 1997; Rasmussen and Collins, 1991). Intensively managed crop or pasture lands has potential for C gain through the use of improved grazing regimes, improved fertilization practices and irrigation management (Follett, 2001; Bruce et al., 1999). We hypothize that increasing plant growth on arid and semiarid lands by conversion to irrigated agriculture is one method that may increase C storage in soils. The objective of this research was to determine if land managed as irrigated moldboard plowed crops converted to irrigated conservation tillage or irrigated pasture could sequester additional C. We use our findings to pose several possible scenarios of altered land management polices that could favor global C sequestration based on the C budgets that we have estimated.

## Materials and Methods

### Site Descriptions

The study area is located on the Snake River Plain, between 42° 30' 00" and 43° 30' 00" N. and 114° 20' 00" and 116° 30' 00" W. The sites occur across an elevational gradient ranging from 860 m to 1300 m. The area is classified as a temperate semi-desert ecosystem (Bailey, 1998). The climate is typified by cool, moist winters and hot, dry summers with annual precipitation ranging from 175 to 305 mm, two-thirds of which occurs during October through March (Collett, 1982). Average annual temperature ranges from 9 to 10 °C. Soils are typically well-drained loams and silt loams derived from loess deposits overlying basalt. Vegetation throughout the general area was historically dominated by basin big sagebrush (*Artemisia tridentata* var. *tridentata* Nutt.), Wyoming big sagebrush (*Artemisia tridentata* var. *wyomingensis* Nutt.), and perennial bunch grasses, including Sandberg bluegrass (*Poa secunda* J. Presl), bottlebrush squirreltail (*Elymus elymoides* Raf. Swezy.), bluebunch wheatgrass (*Pseudoroegneria spicata* Pursh. A. Love), and Thurber's needlegrass (*Achnatherum thurberianum* (Piper) Barkworth).

## Experimental Design

The experiment was arranged in a completely randomized design (Kirk, 1982). Soil samples were taken from: 1) three sites supporting native sagebrush vegetation (NSB) located near agricultural land in Southern Idaho (each site supported a basin big sage and a Wyoming big sage vegetation type) ; 2) three sites that were formerly crop land and converted to and maintained as irrigated pasture (IP) for the past 30 years; 3) three sites that were irrigated crop land and have been managed with conservation tillage (ICT) for the past 8 years; and 4) three irrigated agricultural crop lands in moldboard plowing systems (IMP) that were each growing a) alfalfa, b) wheat, c) potato and d) beans. There were four treatments (NSB, IMP, ICT, and IP) x three sites for each treatment x five cores taken within each treatment at each site (replications) x four soil depths (0-5 cm, 5-15 cm, 15-30 and 30-100 cm) . We took a total of 240 samples.

## Native Vegetation Sagebrush Sites

Native sagebrush sites were vegetated with native steppe vegetation and a low composition of exotic annual grasses. Sites were chosen for this study based on a history of no livestock grazing (BLM, Bruneau Resource Area, unpublished data). All study sites had 5-10 % slope and were on areas that supported basin big sagebrush or Wyoming big sagebrush or communities (Table 1). Soil was classified as a fine, montmorillonitic, mesic Xerollic Haplargid on the Brown's Creek site, a coarse-loamy, mixed non-acid, mesic Xeric Torriorthents on the Simco site and a loamy, mixed, mesic lithic Xerollic Camborthids on the Kuna Butte site (Collett, 1982).

## Irrigated Pasture Sites

Three irrigated pastures were selected that were formerly crop land and converted to and maintained as irrigated pasture (IP) for the past 30 years. The Buhl site was vegetated with Kentucky bluegrass (*Poa pratensis* L.)–orchardgrass.(*Dactylis glomerata* L.) on a Rakane-Blacknest soil complex, fine-loamy, mixed, mesic Xerollic Durargids soil. The Gooding site was vegetated with smooth brome (*Bromus inermis* Leyss.)-orchardgrass on a Paulville-Idow soil complex, fine-loamy, mixed, mesic Xerollic Haplargid soil. The Kimberly site was vegetated with smooth brome-orchardgrass pasture on a Portneuf soil, coarse-silty, mixed, superactive, mesic Durinodic Xeric Haplocalcid soil. Grazing rates on these pastures were 10-12 animal unit months $yr^{-1}$.

**Table 1. Location and characteristics of study sites.**

| Site Name | Latitude Longitude (10' block) | Treatment | Elevation (m) | Site Description | Last Known Fire(s) | Current Dominant Vegetation |
|---|---|---|---|---|---|---|
| Bowns Creek | 43° 20' 00" 115° 56' 00" | Native | 1010 | *Artemisia tridentata* Basin /*Pseudoroegneria* (Pursh) A. Love. *Artemisia tridentata* Wyoming /*Pseudoroegneria* | unknown (>50 years) | *Artemisia tridentata* Basin, Wyoming / *Pseudoroegneria* |

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| Bowns Creek | 43° 20' 00" 115° 56' 00" | Native | 1010 | *Artemisia tridentata* Basin /*Pseudoroegneria* (Pursh) A. Love. *Artemisia tridentata* Wyoming /*Pseudoroegneria* Soil Type: Xeroic Haplargid | unknown (>50 years) | *Artemisia tridentata* Basin, Wyoming / *Pseudoroegneria spicata* / *Poa secunda* |
| Simco | 43° 22' 00" 115° 51 '00" | Native | 1095 | *Artemisia tridentata* Basin /*Pseudoroegneria* *Artemisia tridentata* Wyoming /*Pseudoroegneria* Soil Type: Xeric Torriorthent | unknown (>50 years) | *Artemisia tridentata* Basin, Wyoming / *Pseudoroegneria spicata* / *Poa secunda* |
| Kuna Butte West | 43° 27' 00" 116° 27' 00" | Native | 950 | *Artemisia tridentata* Basin /*Pseudoroegneria* *Artemisia tridentata* Wyoming /*Pseudoroegneria* Soil Type: Xerollic Camborthid | unknown (>50 years) | *Artemisia tridentata* Basin, Wyoming / *Poa secunda* |
| Kimberly Pasture | 42° 00' 00" 114° 30' 00" | Irrigated Pasture | 1210 | *Dactylis glomerata L.* / *Bromus inermis* Lyess. Soil Type: Xeric Haplocalcid | (>100 years) | 50 % *Dactylis glomerata.* 50 % *Bromus inermis* |
| Gooding Pasture | 42° 46' 00" 114 ' 37' 00" | Irrigated Pasture | 1070 | *Bromus inermis* / *Dactylis glomerata* Soil Type: Xerollic Haploargid | (>100 years) | 50 % *Dactylis glomerata* 50 % *Bromus inermis* |
| Buhl Pasture | 42° 30' 00" 114° 30' 00" | Irrigated Pasture | 1073 | *Poa pratensis L.* / *Dactylis glomerata.* Soil Type: Xerollic Durargids | (>100 years) | 50 % *Poa pratensis* 50 % *Dactylis glomerata* |
| Kimberly | 42° 00' 00" 114° 30' 00" | Irrigated moldboard plow | 1260 | Rotations of : *Phaseolus vulgaris L* / *Triticum aestivum L.* / *Solanum tuberosum L* / *Medicago sativa L.* Durinodic Xeric Haplocamberid | (>100 years) | *Phaseolus vulgaris L* / *Triticum aestivum L.* / *Solanum tuberosum L* / *Medicago sativa L.* |
| University of Idaho | 42° 46' 00" 114 ' 37' 00" | Irrigated moldboard plow | 1220 | Rotations of : *Phaseolus vulgaris L* / *Triticum aestivum L.* / *Solanum tuberosum L* / *Mmedicago savtia L.* Durinodic Xeric Haplocalcid | (>100 years) | *Phaseolus vulgaris L* / *Triticum aestivum L.* / *Solanum tuberosum L* / *Mmedicago savtia L.* |
| | | | | | | |
| South Farm | 42° 30' 00" 114° 30' 00" | Irrigated moldboard plow | 1200 | Rotations of : *Phaseolus vulgaris L* / *Triticum aestivum L.* / *Solanum tuberosum L* / *Medicago sativa L.* Durinodic Xeric Haplocalcid | (>100 years) | *Phaseolus vulgaris L* / *Triticum aestivum L.* / *Solanum tuberosum L* / *Medicago sativa L.* |

**Table 1. Continued**

| Site Name | Latitude Longitude (10' block) | Treatment | Elevation (m) | Site Description | Last Known Fire(s) | Current Dominant Vegetation |
|---|---|---|---|---|---|---|
| Kimberly | 42° 00' 00" 114° 30' 00" | Irrigated conservation tillage | 1260 | Rotations of : *Phaseolus vulgaris L / Triticum aestivum L. / Solanum tuberosum L / Medicago sativa L.* Durinodic Xeric Haplocamberid | (>100 years) | *Phaseolus vulgaris L / Triticum aestivum L. / Solanum tuberosum L / Medicago sativa L.* |
| University of Idaho | 42° 46' 00" 114 ' 37' 00" | Irrigated conservation tillage | 1220 | Rotations of : *Phaseolus vulgaris L / Triticum aestivum L. / Solanum tuberosum L / Medicago sativa L.* Durinodic Xeric Haplocalcid | (>100 years) | *Phaseolus vulgaris L / Triticum aestivum L. / Solanum tuberosum L / Medicago sativa L.* |
| South Farm | 42° 30' 00" 114° 30' 00" | Irrigated conservation tillage | 1200 | Rotations of : *Phaseolus vulgaris L / Triticum aestivum L. / Solanum tuberosum L / Medicago sativa L.* Durinodic Xeric Haplocalcid | (>100 years) | *Phaseolus vulgaris L / Triticum aestivum L. / Solanum tuberosum L / Medicago sativa L.* |

## Irrigated Conservation Tillage and Crop Sites

Three sites with fields rotating among alfalfa (*Medicago sativa* L.), wheat (*Triticum aestivum* L.), potato (*Solanum tuberosum* L.) and beans (*Phaseolus vulgaris* L.) were sampled. All sites were located on fields managed by USDA Agricultural Research Service's Northwest Irrigation and Soils Research Laboratory or the University of Idaho, Research and Extension Center. Soil on all sites was classified as a coarse-silty, mixed, superactive, mesic Durinodic Xeric Haplocalcid, with 0.1-0.21 g/g clay and 0.6-0.75 g/g silt, and organic matter of approximately 13 g $kg^{-1}$. The soil has a pH of 7.6-8.0. Slope on these sites ranges from 1.0 to 3.0% (Table 1).

## Sampling Procedures

Soil cores were taken from each site during winter (January), spring (April), summer (August), autumn (November) in 1999. We sampled the top 1 m of soil each season (winter, spring, summer and autumn) to determine if the amount of C in soil would be affected by irrigation, tillage and vegetation. Sampling locations were randomly chosen at each site or field. Separate 10 cm diameter replicate cores were randomly taken and partitioned into 0-5 cm, 5-15 cm, 15-30 and 30-100 cm depths. Roots greater than 1.0 cm diameter were measured separately. Carbon in above-ground vegetation was estimated by measuring the amount of material in 10 separate 1.0 $m^2$ areas in each site or field (Entry and Emmingham, 1998).

## Carbon in Soil and Above-ground Vegetation

Concentration of organic C in each sample of mineral soil was determined by the Walkley-Black procedure and loss on ignition (Nelson and Sommers, 1996). The amount of C per $ha^{-1}$ of the 0-100 cm of mineral soil was calculated assuming 0.44 g C $g^{-1}$ organic matter with correction for soil bulk density. Ten separate 10 cm diameter soil cores were taken to a 1.0 m depth, divided into 0-5 cm, 5-15 cm, 15-30 and 30-100 cm depths to determine bulk density. Bulk density was measured by dividing by the oven dry weight after drying at 105 $^{o}$C for 48 hours by the volume of the sample (Blake and Hartage, 1982). Above-ground vegetation was collected and separated into sage, grass, forbs, herbs and duff. Above-ground material was dried at 80 $^{o}$C for 48 h, weighed and ground to pass a 1 mm opening. Carbon in above-ground vegetation was determined by loss on ignition (Nelson and Sommers, 1996). The amount of C in the above ground material was assumed to contain 0.44 g C $g^{-1}$ organic matter on an ash free basis (Nelson and Sommers, 1996). Calculations estimating C stored in soils in the Western United States and worldwide are based on the Walkley Black procedure.

## Calculations

Concentration of organic C determined by the Walkley-Black procedure was converted, using bulk density measurements, to a meter square basis to a depth of 1 meter. Organic C in kg $m^{-2}$ was converted to Mg C $ha^{-1}$ by multiplying by 10,000 (land area) and dividing by 1,000 (C weight), which is a 1: 10 conversion. We divided the resulting number by 10 to account for a 10 % conversion of one treatment (land area). The amount of C sequestered in the Pacific Northwestern United States, the 11 western states in the United States and worldwide was estimated by multiplying Mg C $ha^{-1}$ by the number of hectares of irrigated land in each area. There are 9,055, 979 ha of land in irrigated crop land in the Pacific Northwest, 24,322,029 ha in the Western U.S. and 260, 000,000 ha worldwide (Bucks et al., 1990; Tribe, 1994; Howell, 2000). The carbon sequestered worldwide ($C_S$) relative to the amount of C projected to be emitted worldwide during the next 30 yr ($C_{EW}$ ) was calculated by dividing the Mg C sequestered in each treatment x area by the total projected worldwide release of $CO_2$ - C during the next 30 years (5.7 x $10^{10}$ Mg C) multiplied by 100.

## Statistical Analysis

All data were subjected to a one way vegetation type analysis of variance (ANOVA) for a completely randomized design (Snedecor and Cochran, 1980; Kirk, 1982). Residuals were normally distributed with constant variance. Statistical Analysis Software programs (SAS Institute Inc. 1996) were used to conduct the analysis of variance. Significance of treatment means were determined at $P < 0.05$ with the Least Square Means test.

# Results and Discussion

## Site Specific Findings

Statistical comparisons in the ANOVA showed that soil bulk density, soil C, site C, net C in soil, net site C in site x vegetation interactions are not significant at $P \leq 0.05$. Therefore, results are discussed with respect to vegetation differences (Snedecor and Cochran 1980; Kirk 1982). Bulk density was greater in soils in IP, ICT and IMP than in NSB soils. Bulk density was less in the NSB 0-5 cm soil depth than the 5-15, 15-30 and 30-100 cm depths and all other soils (Table 2). Soil C was greater in the NSB 0-5 cm soil depth than the 5-15, 15-30 and 30-100 cm depths and all other soils (Table 2). Soil was greater in the in the 0-5 cm, 5-15 cm and 15-30 cm depths in the IP than the IMP or ICT treatments.

Organic C contained in above-ground vegetation was greater on NSB sites than IP; however, IP biomass was removed by grazing. Crops were not considered as permanent vegetation. Prior to adjustment for agricultural $CO_2$ emissions, (total), soil C and C on site was greatest to least in the order IP>ICT>IMP>NSB (Table 3). Conversion of land to IP resulted in less C emitted to the atmosphere than IMP- or ICT- managed crops because less fertilizer and farm operations were necessary. After adjustment for agricultural $CO_2$ emissions, (net) C in soils was greatest to least in the order IP>ICT>NSB>IMP.

We estimated that if NSB sites were converted to IMP a net loss of 0.15 kg C $m^{-2}$ over 30 years would occur (Table 3). We estimated a net gain of 0.80 kg C $m^{-2}$ over 30 years If NSB sites were converted to ICT , and if converted to IP one could expect net gain of 3.56 kg C $m^{-2}$ over a 30 year period. We estimated that if IMP was converted to ICT a net gain of 0.95 kg C $m^{-2}$ over 30 years would occur. If IMP land was converted to IP an estimated net gain of 3.71 kg C $m^{-2}$ over a 30 year period would occur.

**Table 2. Bulk density, Walkley Black C and loss on ignition C (LAI-C) in soils growing native sagebrush, irrigated moldboard plowed cropland, irrigated conservation tilled cropland and irrigated pastures in Southern Idaho.**

| Treatment | Soil Depth | Bulk Density | WBC† | LAI-C |
|---|---|---|---|---|
| | | -- Mg $m^{-3}$ -- | ---------- g C $kg^{-1}$ soil---------- | |
| Native sagebrush | 0-5 | 0.97 b | 12.7 a | 10.1 a |
| | 5-15 | 1.28 ab | 4.7 c | 4.1 c |
| | 15-30 | 1.34 a | 5.4 c | 5.1 c |
| | 30-100 | 1.36 a | 4.0 c | 3.5 c |
| | | | | |
| Irrigated moldboard plow crop | 0-30 ‡ | 1.28 ab | 7.8 b | 6.9 b |
| | 30-100 | 1.37 a | 6.0 bc | 5.1 bc |
| | | | | |
| Irrigated conservation tilled crop | 0-15 ‡ | 1.38 a | 8.9 b | 7.6 b |
| | 15-30 ‡ | 1.38 a | 6.9 bc | 6.8 bc |
| | 30-100 | 1.37 a | 3.7 c | 3.1 c |
| | | | | |
| Irrigated pasture | 0-30 ‡ | 1.33 a | 8.5 b | 7.9 b |
| | 30-100 | 1.40 a | 4.3 c | 3.8 c |

† In each column, values followed by the same letter are not significantly different as determined by the Least Square Means Test (*P* # 0.05), n = 30.

‡ Statistical comparisons in the ANOVA showed that soil bulk density with respect to soil depth were not significant at $P \leq 0.05$. Therefore, data were combined (Snedecor and Cochran 1980; Kirk 1982).

## Regional and Global Implications

Changes in agricultural practices have great potential to sequester C. In most cases converting selected land managed as IMP to ICT or IP can be implemented with modest economic impact to landowners and pose relatively few socioeconomic issues. If agricultural land is managed properly, these practice shifts would also potentially reduce erosion and water or air pollution. Estimating the potential for C sequestration in terrestrial ecosystems is difficult because the dynamics that control C flow among plants, soils and the atmosphere are poorly understood. Storage of C in below ground systems is the best long-term option in terrestrial ecosystems because C in soils has a longer residence time than most plant biomass. Using the values obtained in southern Idaho, we estimated C storage in soils locally, regionally and globally in soils if: 1) 10% of irrigated land now in IMP agriculture was converted back to NSB, 2) all land presently in IMP was converted to ICT and 3) 10 % of land in irrigated IMP was converted to IP. Since increased agricultural production will be necessary to feed an increasing population, it is impractical to suggest that a large portion of land in IMP can be converted to IP

The reported amounts of C stored in native sagebrush vegetation and irrigated agricultural systems are similar throughout the United States and worldwide (Bowman et al., 1999; Collins et al., 1999; Amthor and Huston, 1998; Potter et al., 1998; Rasmussen and Parton, 1994; Schlesinger, 1977;). These data were used to calculate potential C storage for irrigated agriculture in the Pacific Northwestern U.S., the Western U.S., and worldwide over a 30 year period. If land currently in IMP is converted to NSB we estimated a gain of 1.5 Mg C $ha^{-1}$ (Table 4). Little of this land is managed with conservation tillage. We estimate an increase of 9.5 Mg C $ha^{-1}$ over 30 years if the land presently managed with IMP were converted to ICT. Using this value we calculated that $8.6 \times 10^7$ Mg C (0.15% of the total C emitted in the next 30 yr) could potentially be sequestered in irrigated soils in the Pacific Northwestern U.S (Table 4). Using these values to represent C gains for all irrigated crop land in the western U.S. and if land in IMP were converted to ICT, a possible $2.3 \times 10^8$ Mg C (0.40% of the total C emitted in the next 30 yr) could be sequestered in irrigated agricultural soils in the next 30 years. If the world's IMP land were converted to ICT, $2.5 \times 10^9$ Mg C (4.38% of the total C emitted in the next 30 yr) could be sequestered in the next 30 years. A shift of 10 % of current IMP land to ICT is a reasonable conservation practice goal. Similarly, we can not expect all land presently in IMP to be converted to IP, but a 10 % conversion is feasible. If we predict a storage increase of 37.1 Mg C $ha^{-1}$ for IMP converted to IP and assume 10 % of IMP land is converted to IP, an estimated $3.4 \times 10^7$ Mg C (0.05% of the total C emitted in the next 30 yr) could be sequestered in Pacific Northwestern U.S. soils and $9.0 \times 10^7$ Mg C (0.16% of the total C emitted in the next 30 yr) in the Western U.S. soils in the next 30 years. If our study values are used to represent C gains for irrigated crop land worldwide, an estimated $9.6 \times 10^8$ Mg C (1.68% of the total C emitted in the next 30 yr) could be sequestered if irrigated land presently managed as IMP were converted to ICT (Table 4). Conversion of crop land to irrigated pasture could also relieve grazing pressure on public rangelands, an issue of heated debate between environmentalists and ranchers.

If irrigated agriculture is expanded due to increase in water use efficiency, one could expect a gain in C sequestration. If NSB were converted to ICT, 8.0 g C $ha^{-1}$ could be sequestered (Table 4). Predicting a storage increase of 8.0 Mg C $ha^{-1}$ for NSB converted to ICT and assuming 10 % expansion of irrigated agriculture, an estimated $7.2 \times 10^6$ Mg C (0.01% of the total C emitted in the next 30 yr) could be sequestered in Pacific Northwestern U.S. soils and $1.9 \times 10^7$ Mg C (0.033% of the total C emitted in the next 30 yr) in the Western U.S. soils in the next 30 years. If our study values are used to represent C gains for irrigated crop land worldwide, an estimated $2.1 \times 10^8$ Mg C (0.37% of the total C emitted in the next 30 yr) could be sequestered (Table 4). If NSB were converted to IP, 35.6 g C $ha^{-1}$ could be sequestered. Predicting a storage increase of 35.6 Mg C $ha^{-1}$ for NSB converted to IP and assuming 10% expansion of irrigated agriculture, an estimated $3.2 \times 10^8$ Mg C (0.5% of the total C emitted in the next 30 yr) could be sequestered in Pacific Northwestern U.S. soils and $8.7 \times 10^8$ Mg C (1.53% of the total C emitted in the next 30 yr) in the Western U.S. soils in the next 30 years. If our study values are used to represent C gains for irrigated crop land worldwide, an estimated $9.3 \times 10^9$ Mg C (16.3% of the total C emitted in the next 30 yr) could be sequestered

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ed $7.2x10^6$ Mg C
cific Northwestern
yr) in the Western
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tted in the next 30
g C $ha^{-1}$ could be
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Ig C (0.5% of the
tern U.S. soils and
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rigated crop land
: next 30 yr) could

**Table 3. Organic carbon in: soils[1], above ground biomass[2] and on sites at present[3], carbon emitted during agricultural operations[4], net organic carbon in soils[5] and net carbon gain on sites[6].†**

| Vegetation | Carbon Present: Soil[1] ‡ | Carbon Present: Above ground[2] ¶ | Carbon Present: Site[3] | Carbon emitted [4] § | Net Carbon Gain: Soil[5] ‡ | Net Carbon Gain: Site[6] |
|---|---|---|---|---|---|---|
| | kg C $m^{-2}$ | | | | | |
| Native sagebrush | 5.91 c | 0.42 a | 6.34 c | 0.00 d | 5.91 c | 6.34 c |
| Irrigated moldboard plow crops | 7.29 b | 0.00 c | 7.29 b | 1.10 a | 6.19 b | 6.19 c |
| Irrigated conservation till crops | 8.01 b | 0.00 c | 8.01 b | 0.87 b | 7.14 b | 7.14 b |
| Irrigated pasture | 10.14 a | 0.05 b | 10.19 a | 0.29 b | 9.85 a | 9.90 a |

† In each column, values followed by the same letter are not significantly different as determined by the Least Square Means Test (*P* # 0.05), n = 30.
‡ Values of organic C stored in soils are based on the Walkley Black procedure.
¶ Carbon in soils, above ground vegetation and on the sites at the present time.
§ Estimated carbon emitted in production of fertilizer, fuel consumption in farm operations and via irrigation water over a 30-year period.

**Table 4. Potential organic carbon gain by conversion of irrigated lands currently in moldboard plowing systems to conservation tillage, conversion of native sagebrush to irrigated conservation tillage, conversion of native sagebrush to irrigated pasture and conversion of 10% of irrigated lands currently in moldboard plowing systems to irrigated pasture over the next 30 years.**

| Vegetation Conversion | C gained from a 10 % conversion | Pacific Northwest United States ‡ | | Western United States ‡ | | Worldwide ‡ | |
|---|---|---|---|---|---|---|---|
| | –Mg C $ha^{-1}$– | Mg C | % $C_S/C_{EW}$ § | Mg C | % $C_S/C_{EW}$ § | Mg C | %$C_S/C_{EW}$ § |
| Irrigated moldboard plow to irrigated conservation tillage † | 9.5 | $8.6 \times 10^7$ | 0.15 | $2.3 \times 10^8$ | 0.40 | $2.5 \times 10^9$ | 4.38 |
| Native sagebrush to irrigated conservation tillage | 8.0 | $7.2 \times 10^6$ | 0.01 | $1.9 \times 10^7$ | 0.03 | $2.1 \times 10^8$ | 0.37 |
| Native sagebrush to irrigated pasture | 3.56 | $3.2 \times 10^8$ | 0.56 | $8.7 \times 10^8$ | 1.53 | $9.3 \times 10^9$ | 16.32 |
| 10% of irrigated moldboard plow to irrigated pasture † | 37.1 | $3.4 \times 10^7$ | 0.06 | $9.0 \times 10^7$ | 0.16 | $9.6 \times 10^8$ | 1.68 |

† Estimated C gain from 100% conversation of moldboard plow to conservation tillage and 10% conversion of moldboard plow agriculture to irrigated pasture.
‡ Land area in irrigated cropland in Pacific Northwest - 9, 055,979 ha, Western United States 24,322,029 ha, worldwide 260,000,000 ha.
§ %$C_S/C_{EW}$ = carbon sequestered ($C_S$) divided by the amount of C projected to be emitted worldwide during the next 30 yr, which is $5.7 \times 10^{10}$ Mg C ($C_{EW}$) multiplied by 100.

**Table 5. Potential carbon transfer by converting an equal amount (10%) of rainfed moldboard plow land back to native forest or grassland on a basis of 1 unit of irrigated rainfed agricultural land to 1 unit of native forest or grassland and conversion of equal amount (10%) of rainfed moldboard plow land back to native forest or grassland on the basis of 1 unit of irrigated rainfed agricultural land to 2 units of native forest or grassland**

**Table 5. Potential carbon transfer by converting an equal amount (10%) of rainfed moldboard plow land back to native forest or grassland on a basis of 1 unit of irrigated rainfed agricultural land to 1 unit of native forest or grassland and conversion of equal amount (10%) of rainfed moldboard plow land back to native forest or grassland on the basis of 1 unit of irrigated rainfed agricultural land to 2 units of native forest or grassland**

| Conversion of Vegetation | C Stored From Conversion | Pacific Northwest United States | | Western United States | | Worldwide | |
|---|---|---|---|---|---|---|---|
| | Mg C $ha^{-1}$ | Mg C | $\%C_S/C_{EW}$ † | Mg C | $\%C_S/C_{EW}$ † | Mg C | $\%C_S/C_{EW}$ † |
| Rainfed moldboard plow to native forest on a 1 unit:1 unit basis | 5.6 | $5.1 \times 10^7$ | 0.09 | $1.4 \times 10^8$ | 0.24 | $1.5 \times 10^9$ | 2.63 |
| Rainfed moldboard plow to native grassland on a 1 unit:1 unit basis | 13.0 | $1.2 \times 10^8$ | 0.21 | $3.2 \times 10^8$ | 0.56 | $3.4 \times 10^9$ | 5.96 |
| Rainfed moldboard plow to native forest on a 2 unit:1 unit basis | 5.6 | $1.1 \times 10^8$ | 0.18 | $2.8 \times 10^8$ | 0.49 | $3.0 \times 10^9$ | 5.26 |
| Rainfed moldboard plow to native grassland on a 2 unit:1 unit basis | 13.0 | $2.4 \times 10^8$ | 0.42 | $6.4 \times 10^8$ | 1.20 | $6.8 \times 10^9$ | 11.93 |

† $\%C_S/C_{EW}$ = carbon sequestered ($C_S$) divided by the amount of C projected to be emitted worldwide during the next 30 yr, which is $5.7 \times 10^{10}$ Mg C ($C_{EW}$) multiplied by 100.

Since the earth releases 1.9 x $10^9$ Mg C $yr^{-1}$ (Schlesinger, 1995; Amthor and Huston, 1998), the conversion of 10% IMP to IP, resulting in a possible 9.6x$10^8$ Mg C sequestered over 30 yrs, may be insignificant (Table 4). However, if crops were produced via high output irrigated agriculture while less productive rainfed agricultural land were returned to temperate forest or native grassland, an increase of 5.6 and 13 Mg C $ha^{-1}$, respectively, could be gained over 30 years for each unit of rainfed land converted to native vegetation (Table 5). The amount of irrigated agriculture can likely be increased at least 10% solely through increases in irrigation efficiency and waste water reuse (Howell, 2000). Using a conversion basis of 1 unit of irrigated agriculture to return 1 unit of rainfed agricultural land to native forest, if irrigated agriculture were expanded 10% (meaning that an additional 2.6 x $10^7$ ha of arid or semiarid land were irrigated) and the equal amount of land being managed as rainfed agricultural land were converted to native forest, there is potential to sequester 5.1x$10^7$ Mg C (0.09% of the total C emitted in the next 30 yr) in the Pacific North West, 1.41x$10^8$ Mg C (0.24% of the total C emitted in the next 30 yr) in the western U.S. and 1.5x$10^9$ Mg C (2.6% of the total C emitted in the next 30 yr) worldwide (Table 5). If the rainfed agricultural land were converted to native grassland, there is a potential to sequester 1.2x$10^8$ Mg C (0.2% of the total C emitted in the next 30 yr) in the PNW, 3.2x$10^8$ Mg C (0.6% of the total C emitted in the next 30 yr) in the western U.S. and 3.4x$10^9$ Mg C (5.9% of the total C emitted in the next 30 yr) worldwide (Table 5).

However, irrigated agricultural land typically produces twice the crop yield of rainfed agricultural land (Bucks et al., 1990; Howell, 2000). If irrigated agriculture were expanded 10%, each hectare of new irrigated land could produce the same crop yield as 2 ha of rainfed land (Bucks et al., 1990; Tribe, 1994; Howell, 2000). Under this scenario, the conversion of irrigated land to native forest could potentially sequester 1.0x$10^8$ Mg C (0.2% of the total C emitted in the next 30 yr) in the PNW, 2.8x$10^8$ Mg C (0.5% of the total C emitted in the next 30 yr) in the western U.S. and 3.0x$10^9$ Mg C (5.26% of the total C emitted in the next 30 yr) worldwide (Table 5). If converted to native grassland in this 2:1 conversion scenario, there is a potential to sequester 2.4x$10^8$ Mg C (0.4% of the total C emitted in the next 30 yr) in the PNW, 6.4x$10^8$ Mg C (1.2% of the total C emitted in the next 30 yr) in the western U.S. and 6.8x$10^9$ Mg C (11.9% of the total C emitted in the next 30 yr) worldwide. If highly erosive rainfed lands were selected or if rainfed lands urgently needed for habitat restoration were targeted for such a conversion, significant additional erosion, water quality and habitat benefits could also result.

Since native desert or semi-desert has relatively little ecosystem C compared to forest, grassland or wetland ecosystems (Houghton et al., 1999; Amthor and Huston, 1998; Schlesinger, 1977), converting irrigated moldboard plow crop land back to desert or semi-desert could result in a soil C gain of 0.15 kg C $m^{-2}$. A sequestration of only 1.90 x $10^6$ Mg C 30 $yr^{-1}$ might be expected in the Pacific Northwestern U.S., 5.10 x $10^6$ Mg C 30 $yr^{-1}$ in the western U.S. and 5.46 x $10^7$ Mg C 30 $yr^{-1}$ worldwide. This modest C accumulation in soil would be tempered by the fact that the conversion would require substantial policy incentives and decades to implement. Substantially more C may be sequestered by selectively returning rainfed agricultural land derived from forest, grassland or wetlands back to native vegetation. Tropical and temperate forests typically contain from 10-12 kg C $m^{-2}$, grasslands contain from 18-20 kg C $m^{-2}$, and wetland ecosystems contain from 60-70 kg C $m^{-2}$, whereas arid and semi-arid lands contain 5-7 kg C $m^{-2}$ (Houghton et al., 1999; Ross et al., 1999; Schimel et al., 2000). Since nearly a third of the yield and nearly half of the value of crops in the U.S. are produced on irrigated lands predominantly in arid or semi-arid climatic zones (Bucks et al., 1990; Tribe, 1994; Howell, 2000), a strong strategic rationale can be made for expanding irrigated agriculture in these areas for both crop production and C sequestration, if accompanied by selective return of rainfed agricultural land derived from forest, grassland or wetlands back to native vegetation.

## Factors Affecting Interpretation

Grazing affects the quantity and chemical composition of soil organic matter and the distribution of C in the soil profile (Schuman et al., 1999; Frank et al., 1995; Dommar and Williams, 1990). Schuman et al. (1999), Frank et al. (1995) and Dommar and Williams (1990) found that grazing often increases the concentration of soil C. Ecosystems co-evolved with herbivores. The fact that C storage in range and grassland ecosystems may be unaffected, but usually is increased with light grazing, suggests that grazing is an important part of long-term sustainability of these ecosystems (Schuman et al., 1999).

We recognize that the values for potential C gain in our study are estimates. To obtain a more precise estimate of potential C sequestration from management conversions on a worldwide basis it would be necessary to investigate the potential C accumulated in soils in many different vegetation types. Use of these data from Idaho provide an indication of the potential for these kinds of management shifts on a larger scale. Our estimated values for C gain may actually be conservative due to improving land management methods and improving irrigation technology. The C trends that we monitored were the end result of management that predated new technology now available that would have prevented much of the erosion and loss of soil C on our monitored irrigation sites. Most irrigated cropping world-wide uses surface irrigation, with substantial runoff resulting in some transport offsite of C via erosion with sediment and dissolved C in the water.

Flood and furrow irrigation also transport nutrients, pesticides and enteric microorganisms offsite and ultimately to surface and ground water (Sojka and Entry, 2000; Sojka et al., 1998a; 1998b). Conversion of furrow irrigation to sprinkler irrigation reduces C transport off site via sediment and water because of dramatic reductions of offsite flow and leaching (Aase et al., 1998). The use of conservation tillage and improved sprinkler irrigation systems to reduce erosion, especially in combination with new technologies such as the use of polyacrylamide, has potential to further reduce C transport and degradation (Aase et al., 1998). Additional C that may be sequestered resulting from improved long-term inputs of technology needs to be determined to more accurately predict potential C gains by irrigated agriculture in the future. Our estimates made no attempt to adjust C budgets for loss of C due to erosion. Because great improvements in controlling irrigation-induced erosion have occurred in recent years, it is likely that our C storage estimates for irrigated agriculture are conservative.

# Conclusions

As ecosystems mature, they accumulate soil C to a maximum carrying potential, which is controlled by climate, topography, soil type and vegetation (Van Cleve et al., 1993; Dewar, 1991; Harmon et al., 1990). Therefore, at equilibrium, the amount of C added to the soil via vegetation is equal to the amount of C lost through organic matter degradation and other losses (Henderson, 1995) and eventually a maximum limit will be reached. Although irrigated agricultural systems can not accrue C indefinitely, with improved management they can potentially remove substantial amounts of C from the atmosphere for the next 30-50 years. The potential C sequestered on site by conversion of native vegetation to irrigated agriculture is above the steady state equilibrium of native vegetation. This is in contrast to rainfed agricultural systems which are currently attempting modest C gains to attain near- baseline C concentrations by implementing reduced or no-tillage practices. Rainfed agricultural lands with or without no-till practices have soil C values far below those of native vegetation. A third of the yield and nearly half of the value of crops in the U.S. and worldwide are produced on the irrigated 15-17% of arable lands that are largely in arid or semi-arid climatic zones (Tribe, 1994; Bucks et al., 1990). Since the earth releases $1.9 \times 10^9$ Mg C $yr^{-1}$ (Amthor and Huston, 1998; Schlesinger, 1995), the conversion of 10 % IMP to ICT resulting in a possible $2.5 \times 10^9$ Mg C sequestered over 30 years is a modest 4.47% compared to the total C released into the atmosphere over that time period. However, if crops were produced via high output irrigated agriculture, while selected less-productive rainfed agricultural land were returned to temperate forest or native grassland on a world-wide basis, there could be substantial reductions in atmospheric $CO_2$. Policy makers and agricultural research infrastructure should recognize the enormous potential benefit of land and water management strategies, polices and incentives that could expand arid zone irrigated agriculture as a means for efficient food and fiber production along with substantial C sequestration potential. This potential would be enhanced if coupled with selective return of less efficient rainfed agricultural lands derived from forest, grassland or wetlands back to native vegetation. We recognize that such an expansion would have to be accompanied by renewed efforts of water development. While water resource development has been occurring at a modest pace worldwide since 1990, Howell (2000) indicated the potential for increased extent of irrigation via efficiency improvements and waste water use. Recognition of these potential C benefits should provide an incentive to fund research and pursue management strategies that are possible without sacrificing production and which could increase restoration of native ecosystems, reduce erosion and improve water quality through appropriate targeting of the strategy.

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