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Review

Atmospheric carbon mitigation potential of agricultural management in the southwestern USA

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Abstract

Agriculture in the southwestern USA is limited by water supply due to high evaporation and limited seasonal precipitation. Where water is available, irrigation allows for production of a variety of agricultural and horticultural crops. This review assesses the impacts of agriculture on greenhouse gas emission and sequestration of atmospheric C in soils of the hot, dry region of the southwestern USA. In Texas, conservation tillage increased soil organic C by 0.28 Mg C ha^{-1} $year^{-1}$ compared with more intensive tillage. Conversion of tilled row crops to the conservation reserve program or permanent pastures increased soil organic C by 0.32 ± 0.50 Mg C ha^{-1} $year^{-1}$. Soil organic C sequestration was dependent on rotation, previous cropping, and type of conservation tillage employed. Relatively few studies have interfaced management and C cycling to investigate the impacts of grazing management on soil organic C, and therefore, no estimate of C balance was available. Irrigated crop and pasture land in Idaho had soil organic C content 10–40 Mg C ha^{-1} greater than in dryland, native grassland. Soil salinity must be controlled in cropland as soil organic C content was lower with increasing salinity. Despite 75% of the region's soils being classified as calcic, the potential for sequestration of C as soil carbonate has been only scantly investigated. The region may be a significant sink for atmospheric methane, although in general, trace gas flux from semiarid soils lacks adequate characterization. Agricultural impacts on C cycling will have to be better understood in order for effective C sequestration strategies to emerge.
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Keywords: Soil organic carbon; Soil inorganic carbon; No-tillage; Conservation tillage; Irrigation; Livestock production

Contents

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1. Characteristics of the southwestern USA

This chapter covers the south and west portions of the USA classified as having a dry climate, because the potential annual loss of water through evaporation exceeds annual precipitation (Table 1). This region is collectively discussed as the dry domain (Bailey, 1995) and covers approximately 188 Mha or 26.5% of the continental land area of the USA. It is characterized by variable climate, diverse topography and ecosystems, a relatively low, but increasing human population and rapidly changing economic base. The specific ecoregions (Bailey, 1995), originally used by the United States Forest Service, covered in this chapter include the 310 tropical/subtropical steppe (grassland) division (portions of Texas, Oklahoma,

Table 1
Precipitation, temperature, soils and vegetation of the major ecoregions of the southwestern USA

Ecoregion	Area coverage (Mha)	Precipitation (mm)	Temperature (°C)	Dominant order soils	Native vegetation
310: tropical/subtropical steppe division—Arizona, Oklahoma, New Mexico, Texas, Utah	78.7	255–770	4–21	Mollisols, Entisols, Aridisols, Vertisols, Alfisols	Tall grass prairie east—sandsage and bluestem west (TX and OK); mixed grass, woodlands, forests and alpine tundra, xerophytic grasslands mixed with mesquite and cacti (AZ, NM, UT)
320: tropical/subtropical desert division—Arizona, California, New Mexico, Nevada, Texas, Utah	44.7	50–610	10–24	Aridisols, Entisols	Thorny shrubs with sparse shortgrass (AZ, NM, TX); cacti, shrubs, woodlands (CA, AZ, NV)
340: temperate desert division—Idaho, Nevada, Oregon, Utah, Washington	68.9	130–890	3–13	Aridisols, Entisols, Mollisols, Alfisols	Sagebrush gradient to woodlands to fir forests (NV, UT); sagebrush mixed with short grass to juniper woodlands (WA, OR, ID)
260: Mediterranean division—California, Oregon	32.8	150–2550	0–19	Alfisols, Mollisols, Ultisols, Entisols	Sagebrush to chaparral and pinyon to oak to conifers with bunch grasses to redwoods

New Mexico, Arizona, Colorado, and Utah); the 320 tropical/subtropical desert division (portions of Texas, New Mexico, Arizona, California and Nevada); the 340 temperate desert (portions of Colorado, Utah, Nevada, Idaho, Oregon and Washington); and the 260 Mediterranean division (portions of California and Oregon). Agriculture in the region is limited by water supply due to high evaporation and limited seasonal precipitation. Where water is available, irrigation allows for production of a variety of agricultural and horticultural crops.

1.1. Climate

Low annual precipitation, clear skies, and a warm year-round climate are due in part to a quasi-permanent, subtropical, high-pressure ridge over the majority of the region (Sheppard et al., 2002). Generally, environments > 1500 m elevation are classified as semiarid and environments < 900 m elevations are arid. But this simple classification does not represent the climatic variability due to physiographic and topographic relief, rain shadow effects from mountain ranges, and proximity to moisture sources from the Gulf of Mexico, the Gulf of California, and the Pacific Ocean (Sheppard et al., 2002). The determining climatic factor in the region is topography. For example, every 307 m in elevation is equivalent to traveling 480 km north, which provides different habitat for plants and animals. This is illustrated by the transect between Yuma, AZ (30 m above sea level) and the San Francisco Peaks near Flagstaff, AZ (3820 m above sea level) that are separated by 400 km, but are 6000 km apart ecologically. Yuma has a subtropical climate while the tops of the San Francisco Peaks are climatically similar to northern Canada and Alaska (Hendricks, 1985). Elevation results in both temperature and precipitation differences, e.g., Tucson, AZ (1190 m) at the edge of the Sonoran Desert, receives an average of 250 mm of precipitation each year while 60 km distant at the top of Mt. Lemmon (3035 m) receives an average of 900 mm precipitation.

A major feature of summer precipitation in the region is the North American Monsoon (Adams and Comrie, 1997). A monsoonal flow is a distinctive change in wind direction of at least 120°, including mid-tropospheric winds (Ramage, 1971; Bryson and Lowry, 1955). The monsoon brings the majority of the seasonal precipitation (up to 60% of annual total) to this region due to diurnal cycles of surface heating, convection (Sellers and Hill, 1974) and influx of moisture from the Pacific Ocean and the Gulf of Mexico (Wright et al., 2001).

1.2. Soils

In general, most of the southwestern USA west of Texas would fall under the soil order, Aridisols, but due to extreme changes in topography, many soil orders may be present at the smaller scale (Table 2). Mollisols, Vertisols, Alfisols and Entisols are dominant soil orders in Texas and Oklahoma, while Aridisols, Entisols and Mollisols are extensive in New Mexico, Utah, Arizona, Idaho, Oregon, Washington, California and Nevada (Brady, 2000).

Soil depth and soil organic C (SOC) and nitrogen (N) contents are factors of climate, organisms (vegetation and microorganisms), parent material, topography and time of development (Jenny, 1941). Climate is the most influential factor of the five soil forming factors (Jenny, 1980), as temperature and precipitation exert profound effects on the rate of chemical and physical processes.

Soil texture is also important for C retention in warm climates. Nichols (1984) reported that soil clay content had a greater influence on SOC content in 65 soil pedons ($r = 0.90$) than average precipitation ($r = 0.45$) across a range of soils in Oklahoma, Texas and New Mexico. Burke et al. (1989) supported this conclusion with an evaluation of 500 rangeland and cultivated soils, where SOC in native rangeland soils was positively correlated to precipitation, clay content and negatively correlated with temperature. In cultivated fine-textured soils, loss of SOC was limited because of physical protection provided by clay minerals (physical inaccessibility of SOC to microorganisms).

1.3. Vegetation

Annual precipitation determines the type of vegetative cover in rangeland. At low altitude, vegetation consists mostly of creosotebush (*Larrea tridentata*), cacti, sagebrush and sparse annual grasses. At intermediate altitude, mesquite, pinyon (*Pinus* sp.)

Table 2
Soils, crops and ownership of the land resource areas (LRR) of the southwestern USA

LRR	MLRA	Ownership	Suborder soils	Major crops
(I) Southwest plateaus and plains range and cotton—Texas	83D—lower Rio Grande Valley	Private	Ustolls	Cotton, citrus, pasture, winter vegetables
	83ABC—north, west and central Rio Grande Valley	Private	Psamments, Usterts	Sorghum, cotton, small grains with 75–90% rangeland
	81—Edwards Plateau	Private	Ustolls, Ustalfs	Wheat, sorghum, small grains mainly dryland with 85% rangeland
(H) Central Great Plains winter wheat and range —Texas, Oklahoma, New Mexico	80A—central rolling red prairies	Private	Ustolls, Ustalfs	Wheat, oats, cotton and sorghum (20%), with 80% rangeland
	80B—Texas north -central prairies	Private	Ustolls, Ustalfs	Wheat, oats, cotton and sorghum (15%) with 80% rangeland
	78—central rolling red plains	Private	Ustolls, Ustalfs	Winter wheat, sorghum, cotton, alfalfa, peanuts (35%) with 60% rangeland
	77—southern high plains	Private	Ustolls, Ustalfs, Aigids	Dryland wheat, sorghum and cotton (33%), irrigated soybeans, corn, cotton and vegetables (20%) with 40% rangeland
(G) Western Great Plains range and irrigated region—New Mexico	70—Pecos-Canadian plains and valleys	90% private	Orthids, Ustolls	Beans, small grains, sorghum, alfalfa, corn (4%) with 75% rangeland
(D) Western range and irrigated region—Texas, New Mexico, Arizona Utah, Nevada, California, Oregon, Idaho	42—southern desertic basins, plains and mountains	75% private	Orthids, Orthents	Sorghum, alfalfa and forages, cotton (30%) with 70% rangeland
	41—southeastern Arizona basin and range	Private	Argids, Fluvents	Sparse planting of cotton, corn, alfalfa (20%) with 80% rangeland
	40—central Arizona basin and range	Private	Argids, Fluvents	Irrigated citrus, cotton, vegetables (2%) with rest sparse rangeland
	39—Arizona and New Mexico mountains	Private	Ustolls	Timber (80%), corn, alfalfa, wheat, pinto beans
	37—San Juan River Valley mesas and plateaus	75% private	Orthents, Fluvents	50% desert shrub and short grass range; 2% irrigated forage
	36—New Mexico and Arizona plateaus and mesas	80% private	Argids, Fluvents	1% irrigated forage with remainder as rangeland
	35—Colorado and Green River plateaus	Private	Orthents, Orthids	1% irrigated forage with 90% rangeland
	34—central desertic basins, mountains and plateaus	50% private	Orthents, Fluvents	Irrigate pasture (2–5%) with 90% rangeland
	31—Imperial Valley	Private	Argids	Irrigated citrus, winter vegetables, small grains, alfalfa and forages with no range value
	30—Sonoran Basin and range	20% private	Orthents, Argids	Limited rangeland with low forage production

Table 2 (*Continued*)

LRR	MLRA	Ownership	Suborder soils	Major crops
	29—Southern Nevada basin and range	10% private	Argids	Irrigated forages (1%) with limited rangeland
	28A—Great Salt Lake	25% private	Argids	Irrigated forage and small grains, salt playa (65%)
	28B—central Nevada basin and range	10% private	Argids, Orthids	Irrigated alfalfa (1%) with limited rangeland
	27—Fallon-Lovelock	20% private	Orthids	Irrigated forages (1–2%) with limited rangeland
	26—Carson basin and mountain	25% private	Argids	Irrigated alfalfa (2%) with limited rangeland
	25—Owyhee high plateau	25% private	Xerolls	Irrigated forage (2–3%) with limited rangeland
	24—Humboldt area	20% private	Orthids	Irrigated forage (3%) with limited rangeland
	23—Malheur high plateau	25% private	Xerolls	Irrigated forage (1–2%) with limited rangeland
	22—Sierra Nevada range	50% private	Xerults	Pasture, rangeland with 90% forests
	21—Klamath and Shasta valleys and basins	50% private	Xerolls	Irrigated pasture (5–10%) with 90% rangeland
(B) Northwestern wheat and range region–Idaho, Oregon, Washington	13—eastern Idaho plateaus	75% private	Xerolls	Irrigated forage (10%), dry land wheat (25%), with 50% rangeland
	12—Lost River Valley mountains	5% private	Orthids	1% irrigated pasture with 90% forest and rangeland
	11—Snake River plains	50% private	Orthids	Irrigated potatoes, beans, sugar beets, alfalfa (25%) with 50% rangeland
	10—Upper Snake River	40% private	Xerolls	Irrigated potatoes, small grains, and pasture (5%) with 90% rangeland
	8—Columbia plateau	93% private	Xerolls	Dry land wheat, peas, forage (50%) with 40% rangeland
	7—Columbia basin	85% private	Orthids, Xerolls	Dryland wheat (25%), irrigated fruits and forage (15%) with 60% rangeland
(C) California subtropical fruit, truck, and specialty crop region—California	20—southern California mountains	60% private	Xeralfs, Orthents	Urban (5%) with 20% of rangeland grazed
	19—southern California coastal plain	75% private	Xeralfs	Urban (20%) 35% erosion protection, with 10–20% irrigated for subtropicals
	18—Sierra Nevada foothills	80% private	Xerults, Xeralfs	Dry land grains, irrigated fruit, nuts, grapes (5%) with 75% rangeland
	17—Sacramento and San Joaquin Valleys	90% private	Argids, Xeralfs	Cropland with irrigated cotton, fruits, nuts, grapes (50%) with 30% rangeland
	15—central California coast range	80% private	Xeralfs, Xerolls	Dry land wheat (10%) with 85% rangeland
	14—central California coastal valleys	Private	Xeralfs	Urban (20%), 55% irrigated fruits, vegetables, grapes with 25% rangeland

Table 2 (*Continued*)

LRR	MLRA	Ownership	Suborder soils	Major crops
	5—Siskiyou-Trinity area	50% private	Xerults	Grazed, alfalfa, pasture (10%) with 85% forests
	4—California coastal redwood belt	90% private	Humults	Grazing (10%) and forage (3%) with 70% forests

and juniper (*Juniperus* sp.) are dominant. At high altitude, pine (*Pinus* sp.), spruce (*Picea* sp.) and fir (*Abies* sp.) are abundant. Cooler temperature at higher elevation reduces plant litter decomposition and results in an increase in SOC (Kirschbaum, 1995). However, moister condition at higher elevation promoted decomposition compared to warmer, drier sites (Conant et al., 2000). The combined effect of increasing precipitation, change in vegetation and decreasing annual temperature with increasing elevation has been found to increase SOC by 3.2% (relative) per 1000 m of elevation up to 2000 m (Martin and Fletcher, 1943) with an even higher SOC increase in conifer forests > 2000 m (Hendricks, 1985).

One of the debates in climate change scenarios is whether woody species encroachment might cause regional desertification (Bahre, 1991). Desertification is defined as a general shift from forage-productive semiarid grasslands to forage-deficient semiarid savannas (Biggs et al., 2002). The rate and mechanisms involved with woody encroachment are controversial as it has been argued that woody species have always been present in grasslands (Bahre, 1991). Isotopic studies of plant phytoliths (silica–calcium cellular precipitates) in semiarid Arizona grasslands have shown that C_4 grasses dominated the grass composition for the last 8000 years with C_3 grass and woody vegetation increasing only in the last 100–350 years (McClaran and Umlauf, 2000). Other Arizona research has suggested that C_3 trees and shrubs were common in historically C_4 dominated grasslands (Biggs et al., 2002).

Another effect of vegetation on soil development is N fixation and release of N with litter fall. The most studied legume species of the region are mesquite and palo verde (*Cerdicium floridum*) trees. Many studies have found higher soil C, N, K, and S concentrations, and soluble salts in proximity to trees compared to open areas (Tiedemann and Klemmedson, 1973, 1986; Klemmedson and Tiedemann, 1986; Barth and Klemmedson, 1978, 1982). The encroachment of legume species into nutrient limited native grasslands maybe an important factor in C and N cycling. The impacts of legume N inputs on grassland productivity and the C cycle in rangeland may be interrelated with climate (Jackson et al., 2002). Woody encroachment in eastern grasslands (wetter Texas sites) decreased C stock in the upper meter of soil while shrub encroachment in western grasslands (drier New Mexico site) increased C stock (Jackson et al., 2002). Even after removal of the mesquite, Biggs et al. (2002) found that elevated levels of soil nutrients under shrub growth were retained for decades in the SOC. Since the amount of SOC is directly related to soil N content (Martens et al., 2003), any N inputs or losses will influence SOC levels.

1.4. Distribution of land use

Irrigated agriculture currently accounts for nearly 87% of southwestern USA water use (National Synthesis Report, 2001) and an increasing human population will change this distribution of water use (Table 2). Due to low precipitation and the high evapotranspiration potential of the region, coupled with limited surface or ground water supplies, <2% of the region (~3.2 Mha) is presently utilized as irrigated agriculture. The largest agricultural activity in the region is livestock grazing on non-irrigated rangeland. Even in an intensively agricultural state such as California, only about 7.2% of the land is cropland and other non-rangeland agricultural production. Except for sorghum production, row crop production in the southwestern USA has limited potential for sequestration of atmospheric C and N compared with the rest of the USA (USDA-NASS, 2000). The major potential resource of the region for removal of atmospheric C is the area's size (26% of land mass in USA) interacting with range management to sequester SOC and provide a soil sink for atmospheric CH_4.

Rangelands in the region still show the effects of overgrazing by livestock that was prevalent during the late 19th and early 20th centuries. In the 1500s, domestic livestock were introduced by Spanish explorers and small herd of cattle and sheep were distributed through the grasslands of southeastern Arizona, southern New Mexico, western Texas, and coastal California (USDA, 1936; Wagoner, 1952; Burcham, 1957). Fires were frequent (Humphrey, 1958; Hastings and Turner, 1965), and although little is known about the exact composition of vegetation during early settlement, there seems to have been no major changes until present.

Cooperative research began in the 1890s to determine the feasibility of planting forage species to restore land productivity. These early reseeding efforts often failed because most of the species were not adapted to the region. More recent attempts to reintroduce native and improved grass species to improve land and potentially counter shrub encroachment have had limited success (Lavin and Johnsen, 1977; Johnsen and Gomm, 1979; Lavin et al., 1981). With the loss of native seed sources and valuable top soil from erosion, it may not be possible to restore the degraded grasslands.

2. Management impacts on SOC

2.1. Crop management

Crop production in western Texas (~0.75 Mha) is divided into cotton (27%), sorghum (17%), wheat (21%) and corn (10%), totaling 75% of the agricultural commodities produced in Texas on 20% of the land area (USDA-NASS, 2000). Oklahoma has ~0.80 Mha in production with ~98% of production as winter wheat and hay. About 79% of New Mexico's (~0.45 Mha) agricultural production is divided among sorghum, winter wheat and hay. Although, the Welton-Mohawk Valley in Arizona and the Imperial and Coachella Valleys of California have intensive irrigated agriculture, they are very limited in size (<1% of land area in California). Arizona has approximately 0.49 Mha under irrigation with about 47% of the irrigated land in hay or cotton (USDA-NASS, 2000). A large portion of agricultural land in California is in hay, citrus, cotton or grapes and a great deal of the winter produce grown in the USA originate from this region. Nevada has <1% of land area for irrigated hay production to support animal growth (USDA-NASS, 2000). Utah has irrigation on approximately 2.4% of the land area composed of 19% wheat and 70% hay. Idaho has irrigation on 8% of the land area primarily in the Snake River Valley for hay and wheat (65%). Central Washington has about 34% of the land in wheat with the remainder as rangeland.

2.1.1. Conservation tillage

The use of tillage has reduced SOC 25–33% compared with native conditions in the region (Potter et al., 1999). In this review, for studies with SOC reported without bulk density or calculation of SOC on a volumetric basis, bulk density was predicted from a relationship with SOC content (BD = 1.66–0.308% $OC^{0.5}$) as reported by Alexander (1980) from 721 semiarid soils. The impact of different crop rotations and tillage managements on SOC content in the region is listed in Table 3. Research has shown that under reduced or no-tillage management, surface residue C enriches the SOC pool compared with buried residues. Surface placement of crop residues increased soil microbial biomass and activity, increased SOC cycling and nutrient release and resulted in greater SOC in the surface 10–12 cm compared to more intensive tillage. The extent of the increase was dependent on the rotation utilized, previous cropping practices, and type of conservation tillage (no-tillage, ridge till, disk, etc.). From a climatic sequence in Texas, change in SOC between no-tillage and inversion tillage was related to annual temperature as ΔSOC (kg ha^{-1} year^{-1}) = -17.2 [annual temperature (°C)] + 619 ($r^2 = 0.99$) and to annual precipitation as ΔSOC (kg ha^{-1} year^{-1}) = -0.23 [rain (mm)] + 455 ($r^2 = 0.40$) (Potter et al., 1998). No-tillage was more effective at sequestering C in cool and dry climates than in warm and humid areas.

2.1.2. Crop C inputs

Wheat production showed greater potential to increase SOC compared to sorghum, even though sorghum left three times greater crop residue mass at the soil surface (Potter et al., 1998). The use of no-tillage wheat–fallow rotations was not as efficient for stabilizing SOC as more intensive cropping rotations. Increased SOC has an additional benefit in drier

Table 3
Impact of agricultural practice on soil organic C content in land resource regions in the southwestern USA

Reference	MAP/MAT	Soils	Agricultural practice	Duration (years)	Depth (cm)	Soil organic C (Mg ha^{-1})[a]
(I) Southwest plateaus and plains range and cotton region—Texas						
Potter et al. (1998)	660/22	Ochraqualfs	4-year cotton (Co), 4-year corn (C)	15	20	NTC = 21, CPC = 18, MPC = 15
Salinas-Garcia et al. (1997)	765/22	Ochraqualfs	4-year cotton, 4-year corn	16	20	NTCo = 23 CPCo = 18, MPCo = 20
Zibilske et al. (2002)	765/22	Calciustolls	Irrigated cotton–corn rotation	9	12	NT = 20; RT = 20; MP = 17
(H) Central Great Plains winter wheat and range region—Texas, Oklahoma, New Mexico						
Bordovsky et al. (1999)	550/17	Paleustalfs	Rain (R) and irrigated (I), wheat (W), sorghum (S), double crop (DC)	9	15	RRTW = 8, CPW = 10, RTS = 9, CPS = 6, IRTW = 11, CPW = 13, RTS = 12, CPS = 8, IRTDCWS = 15
Dao (1998)	822/17	Paleustolls	Wheat	11	20	NTW = 45, CPW = 42
Gebhart et al. (1994)	430/14	Paleudalfs	Dryland cotton, CRP, native prairie (NP)	5	40	CPC = 18, CRP = 24, NP = 67
Potter and Chichester (1993)	865/19	Pellusterts	Wheat, corn, sorghum	10	25	NT = 44, CP = 39
Potter et al. (1997)	473/14	Paleustolls	Continuous wheat (CW), continuous sorghum (CS)	10	20	NTCW = 33, SMCW = 27, NTCS = 31, SMCS = 28
Potter et al. (1997)	473/14	Paleudalfs	Wheat–sorghum –fallow (WSF)	10	20	NTWSF = 29, SMWSF = 27, NTWF = 28, SMWF=26
Potter et al. (1998)	473/14	Paleustolls	Continuous wheat /sorghum	10	20	NTW = 33, NTS = 31, SMW = 28, SMS = 28
Potter et al. (1998)	860/19	Pellusterts	Wheat/sorghum/corn	6	20	NTC = 46, NTW = 46, CPC = 42, CPW = 47
Potter et al. (1999)	878/20	Haplusterts	Agriculture, grassland native prairie restoration	6, 26, 60	120	Ag = 15.4, pasture = 16.3, prairie = 22.3
Undersander and Reiger (1985)	370/17	Paleustolls	Furrow irrigated wheat-tilled	14	15	Incorporated W = 2.4, Removed W = 2.2, Burned W = 2.1

Table 3 (*Continued*)

Reference	MAP/MAT	Soils	Agricultural practice	Duration (years)	Depth (cm)	Soil organic C (Mg ha^{-1})[a]
Unger (1991)	470/17	Paleustolls	Wheat–sorghum–fallow	7&9	20	NT = 24, SM = 23
Unger (1995)	470/17	Paleustolls	Continuous sorghum	6	4	RT = 6.0, CP = 5.6
(D) Western range and irrigated region—New Mexico, Arizona, Utah, Nevada, Idaho, Oregon and California						
Leavitt et al. (2001)	–/39	Haplargid	Irrigated FACE sorghum	2	15	Initial control = 12.8, 2-year control = 12.0, FACE initial = 12.0, 2-year FACE = 13.4
(B) Northwestern wheat and range region—Idaho, Oregon and Washington						
Douglas et al. (1998)	250–700/7–12	Varied	Native prairie or wheat rainfall gradient	>50	20	Ag < 250 = 15, Ag < 400 = 32, Ag > 500 = 47, NP < 250 = 16, NP < 400 = 44, NP > 500 = 79
Entry et al. (2002)	175–305/9–10	Varied	Native and irrigated alfalfa, wheat, potato, bean	8–30	100	NP = 63, IMP = 73, IRT = 80, IP = 102
Glover et al. (2000)	200/9	Haplocambids	Apple, organic, integrated or conventional (Con)	4	15	Con = 21.3, Integrated = 25.3, organic = 24.2
Karlen et al. (1999)	300/8	Haploxerolls	CRP vs. tilled wheat	4.5–5.5	7.5	CRP = 9.2, W = 8.4 (4.5 years), CRP = 11.7, W = 10.6 (5.5 years)
Mulla et al. (1992)	280/8	Argixerolls	Diverse rotation with. or without alfalfa	>40	15	Rotation + alfalfa = 26.7, R-alfalfa = 18.5
Peterson et al. (2002)	500/9	Haploxerolls	Wheat	20	5	NTW = 19.1, CTW = 16.6
Staben et al. (1997)	260/8	Haploxerolls	CRP vs. tilled wheat	4–7	7.5	CRP = 9.8, CTW = 9.8
Whalen et al. (2000)	400/8	Haploxerolls	Native vs. dryland wheat	37	15	CTW = 31.9, NP = 46.4
(C) California subtropical fruit, truck and specialty crop region—California						
Pratt et al. (1957)		Durixeralfs	Citrus—cover crops fertilizers	29	15	Check = 8.2, urea + cover = 10.0, NO_3^- + cover = 11.8
Martens and Frankenberger (1992)		Durixeralfs	Fallow—organic amendments	2	15	Fallow = 22.0, alfalfa = 23.4, straw = 27.0, sewage sludge = 37.0, poultry manure = 32.3
van Groenigen and van Kissel (2002)		Natrargids	Irrigated cotton	>30/17	10	High salinity = 3.9, low salinity = 9.6
Hu et al. (1997)		Xerofluvents	N fixing cover crops or fertilizers	7	20	Fertilizer = 26.1, N fixing cover = 29.9
Lundquist et al. (1999)		Xerorthents	Organic, low input, or conventional	8	15	Organic = 22.5, low = 20.9, conventional = 17.0

[a] NT, no-tillage; CP, chisel plow; MP, moldboard plow; RT, reduced tillage; SM, stubblemulch; CRP, conservation reserve plan; MAP, mean annual precipitation; MAT, mean annual temperature.

climates to potentially increase soil moisture at planting time. In the drier portions of Texas (MLRA 77-80), yield of sorghum from 1939 to 1997 at Bushland, TX, increased due to better hybrids and implementation of reduced tillage practices that improved soil moisture conditions at planting (Unger and Baumhardt, 1999). Increased yield due to higher spring soil moisture also resulted in greater crop residue mass returned to the soil. No-tillage wheat production appears to have the greatest potential to limit C loss in areas with limited rainfall (250–500 mm total annual precipitation). Improving rotations beyond wheat–fallow to include other crops such as legume species may have great potential to increase SOC (Mulla et al., 1992).

2.1.3. Irrigation

Conversion of native range to irrigated cropland in the Snake River region of Idaho stabilized SOC when crops were managed with limited tillage (Entry et al., 2002). The necessity for soil disturbance to build irrigation furrows and bury cotton residues for pest control suggests that these irrigated lands would require new management concepts to limit tillage if C sequestration were considered. Despite the intensive agricultural output of the San Joaquin Valley of central California, few investigations are available on the impacts of management on SOC. Use of compost or organic amendments may be an option for increasing SOC (Table 3), but adding manure or cover crops with tillage-based irrigation management has limited potential to increase SOC (Pratt et al., 1957; Martens and Frankenberger, 1992; Hu et al., 1997; Lundquist et al., 1999). The heavy dependence on intensive tillage for irrigated agriculture limits SOC stabilization in these croplands.

Salinity management must be a priority in irrigated systems, since a negative relationship was found between soil electrical conductivity level and SOC (van Groenigen and van Kissel, 2002). Increased salinity decreased plant residue mass returned to soil. A major hurdle for implementation of C conservation practices has been the extreme cost of crop production. In California production, cost for strawberries can exceed \$ 40,000 ha^{-1} $year^{-1}$ and management that does not limit disease and pests has not been considered (Husein Ajwa, Vegetable Crops Extension, University of California, personal communication).

2.1.4. Retiring former agricultural land as pasture or set aside

Converting agricultural land to grassland via the conservation reserve program (CRP) has shown potential for sequestering atmospheric C as SOC (Gebhart et al., 1994; Potter et al., 1999; Karlen et al., 1999). Sequestration of C has ranged from 5.0 (Potter et al., 1999) to 11 Mg C ha^{-1} (Gebhart et al., 1994). This positive C change was in addition to the yearly loss of SOC due to tillage-based management (Table 3).

2.2. Livestock grazing management

Although productivity of semiarid and arid rangelands is one–three orders of magnitude lower than forest ecosystems (Ludwig, 1987), rangelands are still productive systems. Measuring current grazing impacts on SOC in the arid/semiarid southwestern USA has been complicated due to the destructive overgrazing that occurred at the end of the 19th century, coupled with drought and erosion, which resulted in a dramatic loss of SOC and at some sites, loss of permanent grass cover.

The impact of livestock grazing on SOC in the southwestern USA is presented in Table 4. A major impact on grassland sustainability in the region has been shrub encroachment. Increased SOC and N in the early stages of mesquite encroachment may be a positive factor for increased grass growth (McClaran and Martens, 2004), but mesquite would fragment the rangeland when tree canopies exclude cattle grazing. Shrub encroachment has potential to increase aboveground biomass production (Asner et al., 2003) and SOC (Geesing et al., 2000). Geesing et al. (2000) reported that SOC increased from 23 (open grass sites adjacent to trees) to 32 Mg C ha^{-1} (20 cm) under mesquite trees. Combining the data from the Geesing et al. (2000) study with other work (Virginia and Jarrell, 1983; Tiedemann and Klemmedson, 1986) suggests an east to west gradient of C accumulation under shrubs across the southwestern USA (Table 4).

Moderate livestock grazing in environments with higher annual precipitation has been reported to increase SOC (Schuman et al., 2002). Unfortunately, the few studies that have investigated grazing management on SOC in semiarid regions have not interfaced management and C cycling, and therefore, no estimate of the C balance is available. Milchunas

Table 4
Impact of rangeland management on soil organic C content in land resource regions in the southwestern USA

Authors	MAP/MAT	Soils	Rangeland practice	Duration (years)	Depth (cm)	Soil organic C (Mg ha^{-1})
(I) Southwest plateaus and plains range and cotton region—Texas						
Geesing et al. (2000)	714/22	Paleustalfs, Argiustolls	Native rangeland	>50	20	Mesquite growth = 32.0, open areas = 23.3
(H) Central Great Plains winter wheat and range region—Texas, Oklahoma, New Mexico						
Asner et al. (2003)	665/17	Paleustalfs	Native pastures with shrubs	63	–	1937 aboveground stock = 3800; 1999 aboveground stocks = 5000
Jackson et al. (2002)	640/17	Paleustolls	Native pastures with mesquite	>70	100	Mesquite = 60.0, open grass = 72.4
Potter et al. (2001)	842/17	Argiustolls	Grazing intensity on loam (L) and silt loam soils (SL)	10	25	SLnotgrazed = 1.9, SLmoderate = 2.7, SLhigh = 2.6; Lnotgrazed = 5.3, Lmoderate = 4.2, Lhigh = 3.7
Teague et al. (2000)	550/17	Paleustalfs	Native pastures (NP), with root plowing (RP)	4, 9, 11, 22	200	RP4 = 28, RP9 = 26, RP11 = 25, RP22 = 22, NP4 = 26, NP9 = 26, NP11 = 26, NP22 = 22
(D) Western range and irrigated region—New Mexico, Arizona, Utah, Nevada, Idaho, Oregon and California						
Barth and Klemmedson (1978)	406/16	Haplargids	Shrub impacted grassland	60	15	Shrub = 0.9, open grass = 0.6
Barth and Klemmedson (1982)	220/16	Haplargids	Shrub impacted semiarid rangeland	>50	60	Mesquite size increased SOC by 1.1 per meter mesquite height
Bird et al. (2002)	247/16	Petrocalcids	Semiarid grassland with shrub growth	>50	10	Mesquite = 46, grass = 28, interspace = 29
Cross and Schlesinger (1999)	220/14	Haplargids	Semiarid grassland and shrub comparison	10	10	Mesquite = 0.24, grass = 0.13
Jackson et al. (2002)	247/16	Petrocalcids	Semiarid grassland	>50	100	Mesquite–shrub = 329, grass = 246
Jackson et al. (2002)	220/14	Haplargids	Semiarid grassland	40	100	Mesquite–shrub = 380, grass = 380
Martens and McLain (2003)	335/16	Haplargid	Riparian and adjacent grassland	>50	60	Riparian vegetation = 23, grassland = 10
Paulsen (1953)	406/16	Haplargids	Semiarid grassland	>50	2.5	Mesquite = 0.19, grass = 0.22
Tiedemann and Klemmedson (1973)	330/16	Haplargids	Semiarid grassland ungrazed	>30	4.5	Mesquite = 0.46, grass = 0.17
Tiedemann and Klemmedson (1986)	330/16	Haplargids	Semiarid grassland ungrazed	>30	19.5	Mesquite = 0.95, grass = 0.53
Virginia and Jarrell (1983)	65/22	Haplargids	Arid grassland	>50	30	Mesquite = 6.9, interspace = 1.4
(B) Northwestern wheat and range region—Idaho, Oregon and Washington						
Chen and Stark (2000)	468/8	Haploxeroll	Rangeland revegetation	15/28A	10	*Agropyron* sp. = 3.8, sagebrush = 3.8, interspace = 3.8

Table 4 (*Continued*)

Authors	MAP/MAT	Soils	Rangeland practice	Duration (years)	Depth (cm)	Soil organic C (Mg ha^{-1})
Svejcar and Sheley (2001)	175/8	Torri-psamment	Rangeland with *B. tectorum*	>40/8	30	Native = 1.5, *B. tectorum* = 1.7
Halvorson et al. (1997)	220/7	Camborthids	Rangeland and 9-year post-fire	>50/7	5	Burned = 0.63, unburned = 0.61
(A) California coastal redwood belt—California and Oregon						
Popenoe et al. (1992)	2000/12	Mollisols	Grassland, conifer-grassland	>50/4	30	Grassland = 14.2, conifer-grassland = 14.1

and Laurenroth (1993) reviewed 97 published articles (276 data sets) in arid and semiarid regions and found that SOC and N responses to grazing were near equally divided between negative and positive impacts, although grazing decreased soil water content at 87% of the sites. Unfortunately, an assessment of soil texture on changes in vegetation and belowground processes was not performed. Potter et al. (2001) reported that SOC increased with grazing on Oklahoma loam soil, while grazing decreased SOC on an adjacent silt loam soil under the same grazing intensity. Moderate grazing intensity in Arizona resulted in a marked decrease in SOC compared to long-term grazing exclosures (Martens and Johnsen, 2002).

Since water is the main factor limiting ecosystem productivity in the southwestern USA, active zones for sequestration of atmospheric C in SOC may be the few remaining major perennial riparian zones (Rio Grande, San Pedro and Colorado Rivers) and interspersed mountain riparian areas (Chambers et al., 1999). Martens and McLain (2003) reported that SOC to a 60 cm depth was 10 Mg C ha^{-1} in adjacent grassland soil and 23 Mg C ha^{-1} in a riparian soil dominated by a mesquite community. A pronounced O horizon was present in the mesquite grove as the use of deeper water sources for growth was not impacted by the long dry periods that limited litter decomposition on the dry soil surface. Thus, the remaining riparian zones may be major sinks for atmospheric C in both standing biomass and SOC.

Potential increase in rangeland SOC may be limited to areas that are not impacted by pinyon/juniper invasion and receive >500 mm precipitation. Biomass production is limited by soil N availability. Microbiotic crusts are a major source of N in the semiarid rangelands (Belnap and Gardner, 1993; Belnap and Harper, 1995), although the increase in soil N content may be utilized by exotic annual plants that outcompete native species (DeFalco et al., 2001).

2.3. Soil carbonate

In the southwestern USA, low rainfall promotes accumulation of soluble salts and carbonate (CO_3^{2-}) near the surface. Globally, arid and semiarid soils contain a large amount of inorganic C in the form of CO_3^{2-} estimated at 750–950 Pg C (Schlesinger, 1985; Eswaran et al., 2000). Based on a map of calcic soils (Machette, 1985), about 75% of the southwestern USA contains inorganic C. Soil inorganic C (SIC) is the third largest global pool behind the oceanic (38,000 Pg C) and soil organic (1550 Pg C) C pools (Schlesinger, 1997). A review of research covering SIC in rangeland was provided by Monger and Martinez-Rios (2001). Soil inorganic C can be classified as lithogenic and pedogenic. Lithogenic CO_3^{2-} is derived from the parent material of soil. Formation of pedogenic CO_3^{2-} requires an arid environment for the precipitation of CO_3^{2-} and Ca/Mg from a non-CO_3^{2-} source. Increasing atmospheric concentration of carbon dioxide (CO_2) may increase soil CO_2 concentration and aid in the formation of SIC. Given the extensive area of calcic soils in the region, CO_3^{2-} deposition could mitigate some of the C being released to the atmosphere by biological processes. Soil inorganic C is controlled by the CO_3^{2-}–HCO_3^- equilibrium:

$$CO_2 + H_2O \Leftrightarrow HCO_3^- + H^+ \tag{1}$$

$$CaCO_3 + H^+ \Leftrightarrow Ca^{2+} + HCO_3^- \tag{2}$$

Increasing soil CO_2 concentration because of higher atmospheric CO_2 or from decomposition of SOC would drive the equations to the right, dissolving $CaCO_3$ and allowing translocation of Ca^{2+} and HCO_3^-. Eqs. (1) and (2) are constantly shifting to the right and left due to changes in environmental conditions that control uptake and loss of soil inorganic C.

Pedogenic CO_3^{2-} accumulates in soil in stages covering thousands of years (Gile, 1970). The turnover rate of soil inorganic C has been estimated at 120 years in New Mexico (Monger and Gallegos, 2000) and at 85,000 years globally (Schlesinger, 1985). Accumulation rates of soil inorganic C have been estimated at 1–120 kg $CaCO_3$-C ha^{-2} $year^{-1}$ (Gile et al., 1981; Marion, 1989; Reheis et al., 1995). With such large variations, these estimates provide only general information to assess climate change mitigation potential with soil inorganic C.

Irrigation can change SIC dynamics. Irrigation water in arid/semiarid regions often contains as much as 1% dissolved CO_2 and when applied to alkaline (higher soil pH) soil, $CaCO_3$ can precipitate. The net CO_2 release via degassing of irrigation water is calculated to return 0.08 Mg C ha^{-1} $year^{-1}$ to the atmosphere.

Irrigation water may not always release CO_2 to the atmosphere. It has been postulated that there is a synergistic relationship between irrigation that produces greater root respiration (shifting Eq. (1) to the right) and leaching of Ca^{2+} and HCO_3^- deeper in the soil profile that precipitates $CaCO_3$ (Sahrawat, 2003). Evidence for this hypothesis has been found in California where irrigation with treated effluent for more than 70 years has produced significant increase in $CaCO_3$ 2–4 m deep within the soil profile (Eshel et al., 2003).

3. Carbon dioxide flux

Carbon dioxide is produced in soil primarily by root respiration and microbial decomposition of SOC. The amount of CO_2 in the Earth's atmosphere is growing by ~3 Pg C per year (Allen et al., 2000), an increase that affects the biosphere directly through its effect on photosynthesis and indirectly through climate change (Houghton et al., 1995). Soil respiration is highly sensitive to temperature and may show a large response to small climatic changes (Schleser, 1982; Schlesinger, 1991; Townsend et al., 1992). While plant and microbial (soil) respiration are controlled by precipitation and temperature in more temperate regions, soil respiration in semiarid ecosystems is largely controlled by SOC pool size and soil moisture, and an increase in temperature could lead to lower soil respiration (Conant et al., 1998, 2000). Thus, in natural semiarid systems, soil respiration increases with both C pool size and mean annual precipitation, but decreases with an increase in mean annual temperature. Improved understanding of the spatial and temporal variations of CO_2 production is needed to accurately quantify annual CO_2 flux in an ecosystem (Fang et al., 1998; Xu and Qi, 2001).

3.1. Cropping system impacts

Tillage, cover type and cropping intensity influence CO_2 flux. A summation of research investigating CO_2 flux from agricultural management in the southwestern USA is given in Table 5. Soil tillage promotes large, short-term CO_2 emission resulting from physical release of gas from soil pores and solution. In support of this, maximum CO_2 emission occurred immediately after tillage and within 2 h of tillage and then decreased to about 20% of maximum in a field study in eastern Texas (Reicosky et al., 1997). Gilmanov et al. (2003) reported that net ecosystem CO_2 uptake in four crop systems in Oklahoma was highest in the tallgrass prairie, at 0.38 Mg CO_2-C ha^{-1} over the growing season. Winter wheat and mixed prairie systems were equal in production, at 0.21 Mg CO_2-C ha^{-1}, while the grazed pasture was the lowest, at 0.17 Mg CO_2-C ha^{-1}. Few reports exist for CO_2 flux in irrigated agricultural systems outside of California (Table 5). One study reported higher CO_2 emission in response to irrigation of cotton in Arizona (Nakayama et al., 1994).

Intensive vegetable production in California typically results in about 10 tillage passes per year (Jackson et al., 2003). Calderon and Jackson (2002) studied the effects of rototilling and disking on CO_2 emission in a vegetable field, and reported that emission was higher immediately after tillage than in the non-tilled control soil. The CO_2 flux was short-lived and lasted <12 h after tillage, confirming the work by Reicosky et al. (1997) that increased CO_2 flux immediately after tillage was due to physical processes (Calderon and Jackson, 2002).

Table 5
Carbon dioxide flux measurements in agricultural and natural systems of ecoregions of the southwestern USA

Reference	Subregion	Cover type	Soil	Treatment	CO_2 analysis method	CO_2 flux (kg CO_2-C ha^{-1} h^{-1})
Region 310: subtropical steppe						
Mielnick and Dugas (2000)	315	Tallgrass prairie (grasses and forbs)	Pellustert in 6-year study	1 burn	IRGA	Soil surface emission = 1.6 over 6-year monitoring period
Suyker et al. (2003)	311	Tallgrass prairie (C_4 grasses)	Argiustoll/ Natrustoll	Burning	Eddy covariance	Ecosystem uptake = 0.2 over 3-year burn/monitoring period. Uptake during daytime (0.7 avg.), emission at nighttime (0.6 avg.)
Reicosky et al. (1997)	315	Bermuda grass	Pellustert	Chisel plowing	Canopy chamber after tillage	Soil surface emission = 13 for 24 h
Reicosky et al. (1997)	315	Sorghum	Pellustert	Chisel plowing	Canopy chamber after tillage	Soil surface emission = 3.6 for 24 h
Gilmanov et al. (2003)	311	Prairie (C_4 grasses)	Argiustoll/ Natrustoll	None	Eddy covariance	Growing season (210 days) ecosystem uptake = 2.8
Gilmanov et al. (2003)	311	Wheat	Argiustolls	None	Eddy covariance	Growing season (210 days) ecosystem uptake = 1.5
Gilmanov et al. (2003)	311	Mixed prairie (grasses and forbs)	Haplustalf	None	Bowen ratio/energy balance	Growing season (210 days) ecosystem uptake = 3.0
Gilmanov et al. (2003)	311	Pasture (grasses and weeds)	Ustiplamment	None	Eddy covariance	Growing season (210 days) ecosystem uptake = 2.6
Region 320: subtropical desert						
Conant et al. (1998)	313	Ponderosa pine	Argiboroll	None	Static absorption	Soil surface emission = 0.5 over 15-month monitoring period
Conant et al. (1998)	313	Pinyon/juniper	Argiustoll/ Argiboroll	None	Static absorption	Soil surface emission = 0.3 over 15-month monitoring period
Conant et al. (1998)	313	Ponderosa pine	Argiboroll	None	Static absorption	Soil surface emission = 0.5 over 15-month monitoring period
Nakayama et al. (1994)	322	Cotton	Torrifluvent	Subsurface drip irrigation	Static chamber	Soil surface emission = 1.5 May–November
Martens and McLain (2003)	322	Mesquite	Torrifluvent	None	Static chamber	Soil surface emission = 0.4 over 1-year monitoring period
Martens and McLain (2003)	322	Sacaton	Torrifluvent	None	Static chamber	Soil surface emission = 0.3 over 1-year monitoring period
Martens and McLain (2003)	322	Annual grasses and forbs	Torrifluvent	None	Static chamber	Soil surface emission = 0.3 over 1-year monitoring period
Emmerich (2003)	321	Desert scrub	Calciorthid	None	Bowen ratio/ energy balance	Ecosystem uptake = 0.1 over 7-year monitoring period. Emission during daytime = 0.1 avg.

Table 5 (*Continued*)

Reference	Subregion	Cover type	Soil	Treatment	CO_2 analysis method	CO_2 flux (kg CO_2-C ha^{-1} h^{-1})
Emmerich (2003)	321	Desert grassland	Calciorthid	None	Bowen ratio/ energy balance	Ecosystem uptake = 0.1 over 7-year monitoring period. Emission during daytime = 0.3 avg.
Region 340: temperate desert						
Leffler et al. (2002)	341	Juniper	Calciorthid	None	IRGA	Soil surface emission = <0.1 over 9-month monitoring period
Region 260: Mediterranean						
Calderon and Jackson (2002)	262	Fallow vegetable field	Xerorthent	Rototilling/ disking	Static chamber	Soil surface emission = 0.1, 0–5 h following tillage
Calderon and Jackson (2002)	262	Fallow vegetable field	Xerorthent	None	Static chamber	Soil surface emission = <0.1 over 5 h
Padgett-Johnson et al. (2003)	262	Grape	Xerorthent	Furrow irrigation	IRGA	Leaf uptake = 4.3 over growing season
Padgett-Johnson et al. (2003)	262	Grape	Xerorthent	Non-irrigated	IRGA	Leaf uptake = 3.0 over growing season
Deverel and Rojstaczer (1996)	261	Asparagus	Histosol overlying clay	None	Static chamber	Soil surface emission = 1.7 over 26-month monitoring period
Deverel and Rojstaczer (1996)	261	Bermuda grass	Histosol overlying sand	None	Static chamber	Soil surface emission = 2.1 over 26-month monitoring period
Deverel and Rojstaczer (1996)	261	Wheat	Histosol overlying clay	None	Static chamber	Soil surface emission = 1.7 over 26-month monitoring period
Wang et al. (1999)	M262	Lemon	NR	Bare soil	IRGA	Soil surface emission = 1.1 over 15-month monitoring period
Wang et al. (1999)	M262	Lemon	NR	Litter-covered soil	IRGA	Soil surface emission = 1.6 over 15-month monitoring period
Wang et al. (1999)	M262	Oak/grassland	NR	None	IRGA	Soil surface emission = 1.6 over 15-month monitoring period

Irrigation has been reported to increase soil CO_2 emission in both tilled and fallow agricultural soils in California (Jackson et al., 2003; Padgett-Johnson et al., 2003). Jackson et al. (2003) reported that the highest CO_2 emission occurred after irrigation in a fallow soil. The 10-fold increase in CO_2 emission after irrigation took 42 h to decline. Carbon dioxide emission from tilled soils was of lower magnitude and shorter duration. Padgett-Johnson et al. (2003) also reported higher CO_2 emission after furrow irrigation compared to non-irrigated controls (Table 5). Drainage from irrigated agriculture causes SOC oxidation and can induce soil subsidence, creating local flood hazards and possibly impacting atmospheric C pools (Rojstaczer and Deverel, 1993). A 30% loss in SOC following conversion to agriculture was reported by Wang et al. (1999) in a central California lemon orchard, where CO_2 emission from bare orchard soil averaged 0.33 kg CO_2-C ha^{-1} h^{-1}.

3.2. Range and pasture impacts

A summation of research on CO_2 flux from rangeland in the southwestern USA is given in Table 5. In the subtropical steppe region, soil moisture is a key factor controlling microbial and plant respiration and plant CO_2 assimilation. In a grazed pasture in Oklahoma, Meyers (2001) reported that midday CO_2 uptake peaked near 2.0 kg CO_2-C ha^{-1} h^{-1}, but that during drought periods, vegetation became inactive and the land surface emitted CO_2 (0.7 kg CO_2-C ha^{-1} h^{-1}). As a result, net ecosystem C fixation was a sink for atmospheric C during non-drought years, but a source in the drought years. Moisture stress also significantly reduced soil CO_2 emission in tallgrass prairie in Texas (Mielnick and Dugas, 2000) and total ecosystem uptake in Oklahoma (Suyker et al., 2003). Suyker et al. (2003) reported net annual exchange of CO_2 near zero during a year with adequate soil moisture, but net emission of CO_2 to the atmosphere during a year with severe moisture stress.

Measurement of SIC change in calcic soils has been limited, but is now occurring. Emmerich (2003) found that calcic sites in Arizona were a source of C to the

Table 6
Nitrous oxide production in soils of agricultural and natural systems in ecoregions of the southwestern USA

Reference	Sub-region	Cover type	Soil	Treatment	Method of analysis	N_2O emission[a] (g N_2O-N ha^{-1} d^{-1} or ng N_2O-N g^{-1} h^{-1})
Region 310: subtropical steppe						
Hutchinson and Brams (1992)	315	Bermuda grass	Paleudalf	Minimum mgmt	Static chambers	~0 for 9-week summer monitoring period
Hutchinson and Brams (1992)	315	Bermuda grass	Paleudalf	Intensive mgmt, 52 kg N ha^{-1}	Static chambers	0–12.0 for 9-week summer monitoring period
Region 320: subtropical desert						
Guilbault and Matthias (1998)	322	Bermuda grass	Haplargid	Effluent water: 30 mg N l^{-1}	Static chambers	17.3–237.6 for 10-week summer monitoring period
Guilbault and Matthias (1998)	322	Desert vegetation	Haplargid	Undisturbed control	Static chambers	0–5.2 for 10-week summer monitoring period
Martens and McLain (2003)	322	Mesquite	Torrifluvent	None	Static chambers	0–7.2 (1-year monitoring period)
Martens and McLain (2003)	322	Sacaton and forbs	Torrifluvent	None	Static chambers	0–2.7 (1-year monitoring period)
Matthias et al. (1993)	322	Bermuda grass	Torrifluvent	50 kg N ha^{-1}	Static chambers	0–34.3 (3-day measurement period)
Matson et al. (1992)	M313	Douglas fir	Torrifluvent	200 kg N ha^{-1} y^{-1}	Static chambers	2.4–12.0 (3-day measurement period)
Matson et al. (1992)	M313	Douglas fir	Torrifluvent	Undisturbed control	Static chambers	0.0–2.4 (3-day measurement period)
Region 260: Mediterranean						
Venterea and Rolston (2000)	262	Alfalfa, tomatoes	Psammaquent	225 μg N g^{-1} soil	Incubations	1.0–12.0 (60-day incubation period)
Venterea and Rolston (2000)	262	Corn, tomatoes	Xerofluvent	580 μg N g^{-1} soil	Incubations	2.0–31.0 (60-day incubation period)
Venterea and Rolston (2000)	262	Annual row crops	Xerorthent	800 μg N g^{-1} soil	Incubations	2.0–12.0 (60-day incubation period)
Hajrasuliha et al. (1998)	262	Seedless grapes	Torriorthent	34 kg N ha^{-1}	Nitrogen balancing	Trace to none
Jackson et al. (2003)	262	Vegetable crops	Xerorthent	Rototilling	Soil microcosms	0–21.6 (14-day incubation period)
Jackson et al. (2003)	262	Vegetable crops	Xerorthent	Non-tilled control	Soil microcosms	0–7.2 (14-day incubation period)

[a] Peak N_2O Flux values for field studies and soil microcosms are given in g N_2O-N ha^{-1} d^{-1}, while N_2O fluxes from laboratory incubations are given in ng N_2O-N g^{-1} h^{-1}.

atmosphere for four consecutive years of measurement.

The environmental impacts of urban growth converting rangeland to cities are of increasing concern in the region. From 1990 to 2000, three Arizona cities doubled in population (USCB, 2000). Urban development could affect the C sink potential. Mean CO_2 uptake of ornamental residential plants ($1.2\ kg\ CO_2$-C $ha^{-1}\ h^{-1}$) was greater than that of the native desert plants ($0.5\ kg\ CO_2$-C $ha^{-1}\ h^{-1}$) in adjacent remnant Sonoran Desert patches (Martin and Stabler, 2002). Martens and McLain (2003) reported that the expansion of mesquite into two diverse semiarid grasslands increased SOC, but the increased C content did not increase CO_2 evolved compared to brushy open sites and grassland sites.

4. Trace gas flux

Mineralization of SOC and N releases compounds to the atmosphere that can contribute to warming of global temperature, the so-called "greenhouse effect". The gases reported to increase global warming, besides water vapor and CO_2, are nitrous oxide (N_2O) and methane (CH_4). While soil processes do not account solely for the global production of greenhouse gases, soils do contribute to the total global trace gas budgets (Follett, 2001).

4.1. Nitrous oxide

Despite the prevalence of fertilized grain production in the subtropical steppe region, little research has been conducted on N_2O emission arising from N amendment to soils. The research on N_2O emission from the southwestern USA is summarized in Table 6. Hutchinson and Brams (1992) studied N_2O emission from N-amended grassland in Texas and reported that emission was stimulated by application of $52\ kg\ N\ ha^{-1}$. Total emission of N_2O under intensively managed (harvested, fertilized) grassland was $0.5\ g\ N\ ha^{-1}\ d^{-1}$ during a 9-week experimental period and $0.2\ g\ N\ ha^{-1}\ d^{-1}$ from unmanaged control plots. The authors hypothesized that the source of N_2O emission resulted from the activity of nitrifying microorganisms, because N oxide emission paralleled the nitrification of applied NH_4^+ and soil water content never exceeded field capacity.

Nitrogen is a limiting factor for plant productivity in arid lands (Ettershank et al., 1978; James and Jurniak, 1978). An exception to this is areas with increasing encroachment of mesquite, a N-fixing leguminous tree that can contribute to N_2O emission (Virginia et al., 1982; Martens and McLain, 2003). Even in N-limited systems rapid N_2O production following rainfall has been reported (Holmes et al., 1996). It has been proposed that N_2O emission immediately following precipitation indicates that denitrifiers have adapted evolutionarily to the highly episodic water availability in semiarid systems (Peterjohn, 1991), although, the mechanism for desiccation tolerance of denitrifying enzymes is currently not known. Guilbault and Matthias (1998) studied N_2O emission from turfgrass irrigated with secondary sewage effluent ($\sim 4\ mg\ NO_3^-$-N L^{-1} and $\sim 14\ mg\ NH_4^+$-N L^{-1}) and found that N_2O emission was highly responsive to soil moisture, averaging $35\ g\ N_2O$-N $ha^{-1}\ d^{-1}$ from turf and $2.1\ g\ N_2O$-N $ha^{-1}\ d^{-1}$ from native vegetation (Table 7). Since the highest N_2O emission coincided with the onset of the North American monsoon, the authors suggested that heterotrophic organisms were responsible for at least some of the N_2O produced in soil.

The unique characteristics of arid soils, including limited soil moisture and lower SOC suggest that nitrification may be relatively more important than denitrification in the production of N_2O in semiarid natural areas (Guilbault and Matthias, 1998). Low SOC concentration should improve the competitive advantage of autotrophic N_2O-producing nitrifiers. Martens and McLain (2003) measured N_2O emission in Arizona on a site dominated by mesquite with $30\ g\ SOC\ kg^{-1}$ and $47\ mg\ NO_3^-$-N kg^{-1} and a grassland site with $18\ g\ SOC\ kg^{-1}$ and $6\ mg\ NO_3^-$-N kg^{-1}. During the monsoon season, N_2O emission averaged $3.2 \pm 2.0\ g\ N_2O$-N $ha^{-1}\ d^{-1}$ from the mesquite site and $0.6 \pm 0.8\ g\ N_2O$-N $ha^{-1}\ d^{-1}$ from the grassland site. The extremely sandy soil limited the formation of anaerobic conditions and N_2O may have been formed during N mineralization. Nitrous oxide emission was also measured during periods of extreme surface soil dryness, indicating that N_2O emission at these sites was a sum of surface and subsurface processes.

Table 7
Methane flux in soils of agricultural and natural systems in ecoregions of the southwestern USA

Reference	Sub-region	Cover type	Soil	Treatment	Method of analysis	CH_4 Flux (g CH_4-C ha^{-1} d^{-1})
Region 320: subtropical desert						
Striegl et al. (1992)	341	Sparse desert vegetation	Sand	Dry season	Static chamber	5.5 uptake March–October with no rainfall for >10 days
Striegl et al. (1992)	341	Sparse desert vegetation	Sand	Monsoon season	Static chamber	14.0 uptake within 10-day following rainfall
Martens and McLain (2003)	322	Grasses and forbs	Torrifluvent	Monsoon season	Static chamber	10.2 uptake during 3-month monitoring period
Martens and McLain (2003)	322	Grasses and forbs	Torrifluvent	Dry season	Static chamber	7.5 uptake during 9-month monitoring period
McLain and Martens (2003)	322	Desert grasses	Torrifluvent	None	Static chamber	5.5 uptake during 1-year monitoring period
McLain and Martens, unpublished data 2002	313	Juniper and pine	Argiustoll	None	Static chamber	15.2 uptake for single measurement,
Region 260: Mediterranean						
Redeker et al. (2000)	262	Rice	Endoaquert	Rice straw incorporation	NR	2252 emission during 90-day monitoring period
Redeker et al. (2000)	262	Rice	Endoaquert	Burnt straw	NR	991 emission during 90-day monitoring period
Redeker et al. (2000)	262	None	Endoaquert	Flooding	NR	435 emission during 90-day monitoring period
Bossio et al. (1999)	262	Rice	Pelloxerert	Rice straw incorporation; winter flood	Static chamber	495 emission during 6-month monitoring period
Bossio et al. (1999)	262	Rice	Pelloxerert	Rice straw incorporation; winter drain	Static chamber	525 emission during 6-month monitoring period
Bossio et al. (1999)	262	Rice	Pelloxerert	Rice straw burn; winter flood	Static chamber	90 emission during 6-month monitoring period
Bossio et al., 1999	262	Rice	Pelloxerert	Rice straw incorporation; winter drain	Static chamber	128 emission during 6-month monitoring period
Fitzgerald et al. (2000)	262	Rice	Endoaquert	Winter flood	Static chamber	736 emission during two-season monitoring period
Fitzgerald et al. (2000)	262	Rice	Endoaquert	Winter drain	Static chamber	758 emission during two-season monitoring period
Lauren et al. (1994)	262	Rice	Haploxeroll	Rice straw incorporation	Static chamber	3618 emission during 6-month growing season
Lauren et al. (1994)	262	Rice	Haploxeroll	Rice straw and purple vetch incorporation	Static chamber	5593 emission during 6-month growing season

In the subtropical desert region, Matthias et al. (1993) reported N_2O emission of 6.3 ± 10.8 g N_2O-N ha^{-1} d^{-1} during a 10 h period following fertilization of sod grass with 50 kg N ha^{-1}. Matson et al. (1992) also found that annual N_2O emission was 0.62 kg N_2O-N ha^{-1} with fertilization in a 50-year-old mixed conifer forest in New Mexico and 0.04–0.07 kg N_2O-N ha^{-1} without fertilizer. Emission of N_2O accounted for 0.35% of the fertilizer N added.

Irrigating fertilized soil has also been shown to increase N_2O emission by forming anoxic microsites where denitrification can occur (Ryden and Lund, 1980; Lowerance et al., 1998; Kessavalou et al., 1998;

Calderon and Jackson, 2002). Venterea and Rolston (2000) performed incubations with different-textured soils and concluded that denitrification was responsible for a significant fraction of the N_2O produced even under well-aerated conditions of 40–42% water-filled pore space. In contrast, Hajrasuliha et al. (1998) studied denitrification after application of NO_3^- and NH_4^+ fertilizers in a trickle-irrigated system in the San Joaquin Valley, and reported that it was difficult to demonstrate denitrification. They hypothesized that the presence of dissolved oxygen in the irrigation water and fast drainage following the termination of irrigation did not permit the development of anaerobic conditions.

Tillage has also been reported to induce the formation of anoxic microsites, thus promoting denitrifier activity. Jackson et al. (2003) reported that denitrification increased significantly after tillage in a silt loam, beginning at day 2 and lasting a week after tillage. They found that increased denitrification after tillage was accompanied by an increase in phosphorylated fatty acids unique for anaerobic *Eubacteria*, suggesting that tillage disrupted internal pore space resulting in anaerobic microsites.

4.2. Methane

Information on CH_4 emission from agricultural management in the southwestern USA is presented in Table 7. Although arid soils would seem unsuitable for CH_4 emission, methanogens can become active, producing significant CH_4 (Peters and Conrad, 1995). McLain and Martens (2003) measured CH_4 emission of 6.6 ± 3.7 g CH_4-C ha^{-1} d^{-1} over a 5-week period in extremely dry Arizona soils prior to monsoon precipitation, although this production probably resulted from termite activity.

Flooded soils during rice production have been identified as an important source of CH_4. The Sacramento Valley, with ~0.2 Mha of rice (Fitzgerald et al., 2000), is the second ranking rice cropping region of the USA, producing 1.67 Tg annually, equivalent to about 0.3% of the world production of rice (Wong, 2003). Emission of CH_4 from flooded rice is primarily dependent upon the availability of C substrates and influenced by soil temperature, texture, and redox state. Net CH_4 emission from the soil surface is also influenced by the extent of bacterial CH_4 uptake within the soil column (Sass et al., 1994; Huang et al., 1998).

With rice cultivation, straw management and flooding can substantially affect CH_4 emission. Several researchers have examined rice straw mana-gement strategies and their effects on net CH_4 production. Redeker et al. (2000) reported that CH_4 emission from straw-incorporated plots was more than twice that from burned plots and nearly five times that from flooded control plots without rice. Bossio et al. (1999) found that CH_4 emission was four to five times higher in straw-incorporated than straw-burned soils. Earlier studies at the same site found that straw incorporation increased the overall size of the microbial community (Bossio and Scow, 1997, 1998), resulting in a faster turnover rate of SOC. Mineralization of C lowered pore space O_2 and promoted the activity of methanogenic bacteria. Fitzgerald et al. (2000) reported that half of annual CH_4 emission from flooded rice occurred during the winter. Although flooding reduced winter CH_4 emission, draining produced a strong flush of CH_4 emission for several days as trapped CH_4 was released from the soil pore space.

Incorporation of straw in flooded rice systems may not necessarily lead to greater greenhouse gas loading when compared to burning, as burning releases CO_2 and a small amount of CH_4 (Fitzgerald et al., 2000). Wong (2003) compared three scenarios for management of rice and reported that (1) CH_4 emission from California rice would be 0.22 Tg CO_2-C equivalent $year^{-1}$ for field burning of rice straw and (2) soil incorporation of rice straw would be 0.92 Tg CO_2-C equivalent $year^{-1}$. However, (3) if rice straw were used to make paper and burned for energy, the CO_2 equivalent of CH_4 release would be only 0.09 Tg $year^{-1}$.

Arid soils may act as strong CH_4 sinks. Striegl et al. (1992) measured uptake of atmospheric CH_4 by soils of the Mojave Desert of 3.29 g CH_4-C ha^{-1} d^{-1}. The CH_4 sink was responsive to soil moisture, tripling during the summer monsoon season. Martens and McLain (2003) found equally strong CH_4 uptake in several ecosystems in Arizona. Potter et al. (1996) suggested that arid/semiarid soils are significant sinks for CH_4, but much work was needed to quantify the importance of the region for promotion or mitigation of potential climate change.

5. Region potential for stabilization of atmospheric C as SOC

Increasing SOC is a greater challenge in warm and dry climates, because the longer warm season would increase SOC loss compared with mesic northern regions. Since crop residues are the inputs for increasing SOC, surface placement of crop residues while limiting soil disturbance is vital to increasing SOC. The potential for SOC stabilization with contrasting management options for agriculture and rangelands in the southwestern USA was described in Tables 3 and 4. These references confirm that the southwestern USA has potential to sequester atmospheric C. The research clearly shows that no-tillage management with more intensive rotations could sequester SOC. Research has suggested that SOC sequestration could be greater with wheat than with sorghum. Irrigated agriculture holds great promise for sequestering SOC, but the entire process of intensive tillage management will have to be better understood in order for effective C sequestration strategies to emerge. The research from Idaho on minimum tilled irrigated agriculture indicates the magnitude of SOC sequestration potential in irrigated agriculture.

6. Key gaps in greenhouse gas mitigation knowledge

Soil processes in semiarid lands have received little research attention, perhaps because of the perceived notion that SOC pools and fluxes are not important on a national or global scale. However, arid and semiarid lands cover as much as one-third of the Earth's surface and their extent may be increasing in response to climate change. The limited amount of row crop agriculture in the southwestern USA emphasizes the importance for understanding rangeland processes and how climate change may impact vegetation and SOC and N dynamics. Since research has shown differences in potential SOC sequestration among crop species, it is important to understand why mineralization and sequestration rates differ. The use of C/N ratios may not be sufficient for understanding biochemical differences of crop residues and their impacts on soil microbial populations, which are important in regulating the C cycle.

Long-term, structured monitoring studies are needed that follow the entire C and N cycle so that a C budget can be made for rangeland and agricultural management options. Research should determine the chemical composition of plant litter and resulting soil mineralization rates with concomitant determination of CO_2 and trace gas emission. This research is needed for all agricultural areas of the southwestern USA, but is vital for the different intensive agricultural areas in the Mediterranean region in California. The impact of livestock grazing on trace gas emission in the semiarid regions is not well understood. An immediate need for well designed, grazing management studies to estimate trace gas emission and SOC change is of extreme importance.

Recent studies confirm that the soils of the southwestern USA contribute to an increase in atmospheric CO_2 and N_2O concentrations, and mitigate atmospheric CH_4. Agricultural practices impact the extent and direction of these trace gas fluxes. At the same time, these studies have not clarified the uncertainty in regional and national estimates of trace gas exchange. As techniques and models for quantifying gas flux improve, further study is warranted to reduce the uncertainties in the regional, national, and global budgets for CO_2, N_2O, and CH_4.

7. Challenge for the immediate future

Atmospheric trace gas fluxes from semiarid soils are one of the significant unknowns in the global C budget. Clearly, more information is needed to characterize CH_4 emission and uptake potential of soil. The lack of data on N_2O emission and management effects on SOC in California agriculture needs to change. An additional area of needed research is to understand the contribution of inorganic C flux to the C budget. With 75% of soils in the southwestern USA classified as containing carbonates, understanding potential changes in kinetics with changes in temperature and precipitation will be extremely important for completing a C balance for the region.

References

Adams, D.K., Comrie, A.C., 1997. The North American monsoon. Bull. Am. Meteorol. Soc. 78, 2197–2213.

Alexander, E.B., 1980. Bulk densities of California soils in relation to other soil properties. Soil Sci. Soc. Am. J. 44, 689–692.

Allen, A.S., Andrews, J.A., Finzi, A.C., Matamala, R., Richter, D.D., Schlesinger, W.H., 2000. Effects of free-air CO_2 enrichment (face) on belowground processes in a *Pinus taeda* forest. Ecol. Appl. 10, 437–448.

Asner, G.P., Archer, S., Hughes, R.F., Ansley, R.J., Wessman, C.A., 2003. Net changes in regional woody vegetation cover and carbon storage in Texas drylands, 1937–1999. Glob. Change Biol. 9, 316–335.

Bahre, C.J., 1991. Legacy of Change: Historic, Human Impact on Vegetation in the Arizona Borderlands. University of Arizona Press, Tucson.

Bailey, R.G., 1995. Description of the Ecoregions of the United States, Misc. Publ. 1391, 2nd ed. United States Department of Agriculture, Forest Service, 108 pp.

Barth, R.C., Klemmedson, J.O., 1978. Shrub-induced spatial patterns of dry matter, nitrogen, and organic carbon. J. Soil Sci. Soc. Am. 42, 804–809.

Barth, R.C., Klemmedson, J.O., 1982. Amount and distribution of dry matter, nitrogen and organic carbon in soil–plant systems of mesquite and palo verde. J. Range Manage. 35, 412–418.

Belnap, J., Gardner, J.S., 1993. Soil microstructure in soils of the Colorado Plateau: the role of the cyanobacterium *Microcoleus vaginatus*. Great Basin Nat. 53, 362–367.

Belnap, J., Harper, K.T., 1995. Influence of crypotobiotic soil crusts on elemental content of tissue of two desert seed plants. Arid Soil Res. Rehab. 9, 107–115.

Biggs, T.H., Quade, J., Webb, R.H., 2002. $\delta^{13}C$ values of soil organic matter in semiarid grasslands with mesquite (*Prosopis*) encroachment in southeastern Arizona. Geoderma 110, 109–130.

Bird, S.B., Herrick, J.E., Wander, M.M., Wright, S.F., 2002. Spatial heterogeneity of aggregate stability and soil carbon in semiarid rangeland. Environ. Pollut. 116, 445–455.

Bordovsky, D.G., Choudhary, M., Gerard, C.J., 1999. Effect of tillage, cropping and residue management on soil properties in the Texas rolling plains. Soil Sci. 164, 331–340.

Bossio, D.A., Scow, K.M., 1997. Management changes in rice production alter microbial community. Calif. Agric. 51, 33–40.

Bossio, D.A., Scow, K.M., 1998. Impacts of carbon and flooding on soil microbial communities: phospholipid fatty acid profiles and substrate utilization patterns. Microb. Ecol. 35, 265–278.

Bossio, D.A., Horwath, W.R., Mutters, R.G., van Kessel, C., 1999. Methane pool and flux dynamics in a rice field following straw incorporation. Soil Biol. Biochem. 31, 1313–1322.

Brady, N.C., 2000. Elements of the Nature and Properties of Soils. Prentice-Hall, New Jersey, p. 559.

Bryson, R.A., Lowry, W.P., 1955. Synoptic climatology of the Arizona summer precipitation singularity. Bull. Am. Meteorol. Soc. 36, 329–339.

Burcham, L.T., 1957. California Rangeland: An Historico-ecological Study of the Range Resource of California. Div. For., Dept. Nat. Resources, Sacramento, CA, 261 pp.

Burke, I.C., Yonker, C.M., Parton, W.J., Cole, C.V., Flach, K., Schimel, D.S., 1989. Texture, climate and cultivation effects on soil organic matter content in U.S. grassland soils. Soil Sci. Soc. Am. J. 53, 800–805.

Calderon, F.J., Jackson, L.E., 2002. Rototillage, disking, and subsequent irrigation: effects on soil nitrogen dynamics, microbial biomass, and carbon dioxide efflux. J. Environ. Qual. 31, 752–758.

Chambers, J.C., Blank, R.R., Zamudio, D.C., Tausch, R.J., 1999. Central Nevada riparian areas: physical and chemical properties of meadow soils. J. Range Manage. 52, 92–99.

Chen, J., Stark, J.M., 2000. Plant species effects and carbon and nitrogen cycling in a sagebrush-crested wheatgrass soil. Soil Biol. Biochem. 32, 47–57.

Conant, R.T., Klopatek, J.M., Malin, R.C., Klopatek, C.C., 1998. Carbon pools and fluxes along an environmental gradient in northern Arizona. Biogeochemistry 43, 43–61.

Conant, R.T., Klopatek, J.M., Klopatek, C.C., 2000. Environmental factors controlling soil respiration in three semiarid ecosystems. Soil Sci. Soc. Am. J. 64, 383–390.

Cross, A.F., Schlesinger, W.H., 1999. Plant regulation of soil nutrient distribution in the northern Chihuahuan desert. Plant Ecol. 145, 11–25.

Dao, T.H., 1998. Tillage and crop residue effects on carbon dioxide evolution and carbon storage in a Paleustoll. Soil Sci. Soc. Am. J. 62, 250–256.

DeFalco, L.A., Detling, J.K., Tracy, C.R., Warren, S.D., 2001. Physiological variations among native and exotic winter annual plants associated with microbiotic crusts in the Mojave Desert. Plant Soil 234, 1–14.

Deverel, S.J., Rojstaczer, S., 1996. Subsidence of agricultural lands in the Sacramento-San Joaquin Delta, California: role of aqueous and gaseous carbon fluxes. Water Resour. Res. 32, 2359–2367.

Douglas, C.L., Rasmussen, P.E., Collins, H.P., Albrecht, S.L., 1998. Nitrogen mineralization across a climosequence in the Pacific Northwest. Soil Biol. Biochem. 30, 1765–1772.

Emmerich, W.E., 2003. Carbon dioxide fluxes in a semiarid environment with high carbonate soils. Agric. For. Meteorol. 116, 91–102.

Entry, J.A., Sojka, R.E., Shewmaker, G.E., 2002. Management of irrigated agriculture to increase organic carbon storage in soils. Soil Sci. Soc. Am. J. 66, 1957–1964.

Eshel, G., Fine, P., Levy, G.J., Mingelgrin, U., Singer, M.J., 2003. Enhancing inorganic carbon sequestration by irrigation management. Abstract. Paper No. 78-6. Geological Society of America Meeting, Reno, July 2003.

Eswaran, H., Reich, P.F., Kimble, J.M., Beinroth, F.H., Padmanabhan, E., Moncharoen, P., 2000. Global carbon sinks. In: Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A. (Eds.), Global Climate Change and Pedogenic Carbonates. CRC Press, Boca Raton, FL, pp. 15–26.

Ettershank, G., Ettershank, J., Bryant, M., Whitford, W.G., 1978. Effects of nitrogen fertilization on primary production in a Chihuahuan desert ecosystem. J. Arid Environ. 1, 135–139.

Fang, C., Moncrief, J.B., Gholz, H.L., Clark, K.L., 1998. Soil CO_2 efflux and its spatial variation in a Florida slash pine plantation. Plant Soil 205, 135–146.

Fitzgerald, G.J., Scow, K.M., Hill, J.E., 2000. Fallow season straw and water management effects on methane emissions in California rice. Glob. Biogeochem. Cycl. 14, 767–776.

Follett, R.F., 2001. Soil management concepts and carbon sequestration in cropland soils. Soil Till. Res. 61, 77–92.

Geesing, D., Felker, P., Bingham, R.L., 2000. Influence of mesquite (*Prosopis grandulosa*) on soil nitrogen and carbon development: implications for global carbon sequestration. J. Arid Environ. 46, 157–180.

Gebhart, D.L., Johnson, H.A., Mayeux, H.S., Polley, H.W., 1994. The CRP increases soil organic carbon. J. Soil Water Conserv. 49, 488–492.

Gile, L.H., 1970. Soils of the Rio Grande Valley Border in southern New Mexico. Soil Sci. Soc. Am. Proc. 34, 465–472.

Gile, L.H., Hawley, J.W., Grossman, R.B., 1981. Soils and Geomorphology in the Basin and Range Area of Southern New Mexico—Guidebook to the Desert Project, Memoir 39. New Mexico Bureau of Mines and Mineral Resources, Socorro, NM, p. 222.

Gilmanov, T.G., Verma, S.B., Sims, P.L., Meyers, T.P., Bradford, J.A., Burba, G.G., Suyker, A.E., 2003. Gross primary production and light response parameters of four southern plains ecosystems estimated using long-term CO_2-flux tower measurements. Glob. Biogeochem. Cycl. 17, 1–16.

Glover, J.D., Reganold, J.P., Andrews, P.K., 2000. Systematic method for rating soil quality of conventional, organic, and integrated orchards in Washington State. Agric. Ecol. Environ. 80, 29–45.

Guilbault, M.R., Matthias, A.D., 1998. Emissions of N_2O from Sonoran Desert and effluent-irrigated grass ecosystems. J. Arid Environ. 38, 87–98.

Hajrasuliha, S., Rolston, D.E., Louie, D.T., 1998. Fate of ^{15}N fertilizer applied to trickle-irrigated grapevines. Am. J. Enol. Vitic. 49, 191–198.

Halvorson, J.J., Bolton Jr., H., Smith, J.L., 1997. The pattern of soil variables related to *Artemisia tridentate* in a burned shrub-steppe site. Soil Sci. Soc. Am. J. 61, 287–294.

Hastings, J.R., Turner, R.M., 1965. The Changing Mile. Univ. Arizona Press, Tucson, p. 317.

Hendricks, D.M., 1985. Arizona Soils. University of Arizona Press, Tucson, AZ.

Holmes, R.M., Jones Jr., J.B., Fisher, S.G., Grimm, N.B., 1996. Denitrification in a nitrogen-limited stream ecosystem. Biogeochemistry 33, 125–146.

Houghton, J.T., Meira Filho, L.G., Callander, B.A., Harris, N., Kattenberg, A., Maskell, K., 1995. Climate Change: The Science of Climate Change. Cambridge Univ. Press, Cambridge, UK.

Hu, S., Grunwald, N.J., van Bruggen, A.H.C., Gamble, G.R., Drinkwater, L.E., Shennan, C., Demment, M.W., 1997. Short-term effects of cover crop incorporation on soil carbon pools and nitrogen availability. Soil Sci. Soc. Am. J. 61, 901–911.

Huang, Y., Sass, R.L., Fisher, F.M., 1998. A semi-empirical model of methane emission from flooded rice paddy soils. Glob. Change Biol. 4, 247–268.

Humphrey, R.R., 1958. The desert grassland—a history of vegetational change and an analysis of causes. Bot. Rev. 14, 193–252.

Hutchinson, G.L., Brams, E.A., 1992. NO versus N_2O emissions from an NH_4^+-amended Bermuda grass pasture. J. Geophys. Res. 97 (D9), 9889–9896.

Jackson, R.B., Banner, J.L., Jobbagy, E.G., Pockman, W.T., Wall, D.H., 2002. Ecosystem carbon loss with woody plant invasion of grasslands. Nature 418, 623–626.

Jackson, L.E., Calderón, F.J., Steenwerth, K.L., Scow, K.M., Rolston, D.E., 2003. Responses of soil microbial processes and community structure to tillage events and implications for soil quality. Geoderma 114, 305–317.

James, D.W., Jurniak, J.J., 1978. Nitrogen fertilization of dominant plants in the northeastern Great Basin Desert. In: West, N.E., Skujins, J. (Eds.), Nitrogen in Desert Ecosystems. Dowden, Hutchinson, and Ross, Stroudsberg, pp. 219–231.

Jenny, H., 1941. Factors of Soil Formation. McGraw-Hill, New York, p. 281.

Jenny, H., 1980. The Soil Resource: Origin and Behavior. Ecological Studies, vol. 37. Springer, New York.

Johnsen Jr., T.N., Gomm, F.B., 1979. Forage plantings on six Arizona pinyon–juniper subtypes. J. Range Manage. 34, 131–136.

Karlen, D.L., Rosek, M.K., Gardner, J.C., Allan, D.L., Alms, M.J., Bezdicek, D.F., Flock, M., Huggins, D.R., Miller, B.S., Staben, M.L., 1999. Conservation reserve program effects on soil quality indicators. J. Soil Water Conserv. 54, 439–444.

Kessavalou, A., Doran, J.W., Mosier, A.R., Drijber, R.A., 1998. Greenhouse gas fluxes following tillage and wetting in a wheat–fallow cropping system. J. Environ. Qual. 27, 1005–1116.

Kirschbaum, M.U.F., 1995. The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic C storage. Soil Biol. Biochem. 27, 735–760.

Klemmedson, J.O., Tiedemann, A.R., 1986. Long-term effects of mesquite removal on soil characteristics. II. Nutrient availability. Soil Sci. Soc. Am. J. 50, 476–480.

Lauren, J.G., Pettygrove, G.S., Duxbury, J.M., 1994. Methane emissions associated with a green manure amendment to flooded rice in California. Biogeochemistry 24, 53–65.

Lavin, F., Johnsen Jr., T.N., 1977. Species adapted for planting Arizona pinyon–juniper woodland. J. Range Manage. 30, 410–415.

Lavin, F., Johnsen Jr., T.N., Gomm, F.B., 1981. Mulching, furrowing and fallowing of forage plantings on Arizona pinyon–juniper ranges. J. Range Manage. 34, 171–177.

Leavitt, S.W., Pendall, E., Paul, E.A., Brooks, T., Kimball, B.A., Pinter Jr., P.J., Johnson, H.B., Matthias, A., Wall, G.W., LaMorte, R.L., 2001. Stable-carbon isotopes and soil organic carbon in wheat under CO_2 enrichment. New Phytol. 150, 305–314.

Leffler, A.J., Ryel, R.J., Hipps, L., Ivans, S., Caldwell, M.M., 2002. Carbon acquisition and water use in a northern Utah *Juniperus osteosperma* (Utah juniper) population. Tree Physiol. 22, 1221–1230.

Lowerance, R., Johnson Jr., J.C., Newton, G.L., Williams, R.G., 1998. Denitrification from soils of a year-round forage production system fertilized with liquid dairy manure. J. Environ. Qual. 27, 1504–1511.

Ludwig, J.A., 1987. Primary production in arid lands: myths and realities. J. Arid Environ. 13, 1–7.

Lundquist, E.J., Scow, K.M., Jackson, L.E., Uesugi, S.L., Johnson, C.R., 1999. Rapid response of soil microbial communities from conventional, low input and organic farming systems to a wet/dry cycle. Soil Boil. Biochem. 31, 1661–1675.

Machette, M.N., 1985. Calcic soils of the southwestern United States. In: Weide, D.L. (Ed.), Soils and Quaternary Geology of the Southwestern United States, Geological Soc. Am. Spec. Paper 203. Geological Soc. Am., Boulder, CO, pp. 1–21.

Marion, G.M., 1989. Correlation between long-term pedogenic $CaCO_3$ formation rate and modern precipitation in deserts of the American southwest. Quat. Res. 32, 291–295.

Martens, D.A., Frankenberger Jr., W.T., 1992. Modification of infiltration rates in an organic-amended irrigated soil. Agron. J. 84, 707–717.

Martens, D.A., Johnsen Jr., T.N., 2002. Options for management of rangeland carbon and nitrogen content, Agron. Abstr. CD, s07-martens-134850-P.

Martens, D.A., McLain, J.E.T., 2003. Vegetation community impacts on soil carbon, nitrogen, and trace gas fluxes. In: Proceedings of the First Interagency Conference on Research in the Watersheds, Benson, AZ, October 27–30. pp. 521–528.

Martens, D.A., Reedy, T.E., Lewis, D., 2003. Soil organic carbon content and composition of 130-year crop, pasture and forest land-use managements. Glob. Change Biol. 10, 65–78.

Martin, C.A., Stabler, L.B., 2002. Plant gas exchange and water status in urban desert landscapes. J. Arid Environ. 51, 235–254.

Martin, W.P., Fletcher, J.E., 1943. Vertical zonation of great soil groups on Mt. Grahm, Arizona, as correlated with climate, vegetation, and profile characteristics. Arizona Agric. Exp. Stat. Tech. Bull. 99, 65.

Matson, P.A., Gower, S.T., Volkmann, C., Billow, C., Grier, C.C., 1992. Soil nitrogen cycling and nitrous oxide flux in a Rocky Mountain Douglas-fir forest: effects of fertilization, irrigation and carbon addition. Biogeochemistry 18, 101–117.

Matthias, A.D., Artiola, J.F., Musil, S.A., 1993. Preliminary study of N_2O flux over irrigated bermudagrass in a desert environment. Agric. For. Meteorol. 64, 29–45.

McClaran, M.P., Umlauf, M., 2000. Desert grassland dynamics estimated from carbon isotopes in grass phytoliths and soil organic matter. J. Veg. Sci. 11, 71–76.

McClaran, M.P., Martens, D.A., 2004. Soil carbon differences beneath open grassland, and old, new and cut mesquite trees in the desert grassland. Abstract. Society for Range Management Annual Meetings, Salt Lake City, UT.

McLain, J.E.T., Martens, D.A., 2003. Methane consumption and production in desert ecosystems experiencing mesquite invasion and control. EOS Trans. AGU, 84 (46), Fall Meet. Suppl., Abstract B32B-0379.

Meyers, T.P., 2001. A comparison of summertime water and CO_2 fluxes over rangeland for well watered and drought conditions. Agric. For. Meteorol. 106, 205–214.

Mielnick, P.C., Dugas, W.A., 2000. Soil CO_2 flux in a tallgrass prairie. Soil Biol. Biochem. 32, 221–228.

Milchunas, D.G., Laurenroth, W.K., 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. Ecol. Monogr. 63, 327–366.

Monger, H.C., Gallegos, R.A., 2000. Biotic and abiotic processes and rates of pedogenic carbonate accumulation in the southwestern United States—relationship to atmospheric CO_2 sequestration. In: Lal, R., Kimble, J.M., Eswaran, H., Stewart, B.A. (Eds.), Global Climate Change and Pedogenic Carbonates. CRC Press, Boca Raton, FL, pp. 273–289.

Monger, H.C., Martinez-Rios, J.J., 2001. Inorganic carbon sequestration in grazing lands. In: Follett, R.F., Kimble, J.M., Lal, R. (Eds.), The Potential of U. S. Grazing Lands to Sequester Carbon and Mitigate the Greenhouse Effect. Lewis Publishers, Boca Raton, FL, pp. 87–118.

Mulla, D.J., Huyck, L.M., Raganold, J.P., 1992. Temporal variations in aggregate stability on conventional and alternative farms. Soil Sci. Soc. Am. J. 56, 1620–1624.

National Synthesis Report, 2001. Climate change impacts on the United States: the potential consequences of climate variability and change. Report for the US Global Change Research Program. Cambridge University Press, Cambridge, UK, p. 620.

Nakayama, F.S., Huluka, G., Kimball, B.A., Lewin, K.F., Nagy, J., Hendrey, G.R., 1994. Soil carbon dioxide fluxes in natural and CO_2-enriched systems. Agric. For. Meteorol. 70, 131–140.

Nichols, J.D., 1984. Relation of organic carbon to soil properties and climate in the southern Great Plains. Soil Sci. Soc. Am. J. 48, 1382–1384.

Padgett-Johnson, M., Williams, L.E., Waler, M.A., 2003. Vine water relations, gas exchange, and vegetative growth of seventeen *Vitis* species grown under irrigated and nonirrigated conditions in California. J. Am. Soc. Hort. Sci. 128 (2), 269–276.

Paulsen Jr., H.A., 1953. A comparison of surface soil properties under mesquite and perennial grass. Ecology 34, 727–732.

Peterjohn, W.T., 1991. Denitrification: enzyme content and activity in desert soils. Soil Biol. Biochem. 23 (9), 845–855.

Peters, V., Conrad, R., 1995. Methanogenic and other strictly anaerobic bacteria in desert soil and other oxic soils. Appl. Environ. Microbiol. 61, 1673–1676.

Peterson, S.O., Frohne, P.S., Kennedy, A.C., 2002. Dynamics of a soil microbial community under spring wheat. Soil Sci. Soc. Am. J. 66, 826–833.

Popenoe, J.H., Bevis, K.A., Gordon, B.R., Sturhan, N.K., Hauxwell, D.L., 1992. Soil–vegetation relationships in Franciscan terrain of northwestern California. Soil Sci. Soc. Am. J. 56, 1951–1959.

Potter, C.S., Matson, P.A., Vitousek, P.M., Davidson, E.A., 1996. Process modeling of controls on nitrogen trace gas emissions from soils worldwide. J. Geophys. Res.-Atm. 101, 1361–1377.

Potter, K.N., Chichester, F.W., 1993. Physical and chemical properties of a vertisol with continuous controlled-traffic, no-till management. Trans. ASAE 36, 95–99.

Potter, K.N., Jones, O.R., Torbert, H.A., Unger, P.W., 1997. Crop rotation and tillage effects on organic carbon sequestration in the semiarid southern Great Plains. Soil Sci. 162, 140–147.

Potter, K.N., Torbert, H.A., Jones, O.R., Matocha, J.E., Morrison Jr., J.E., Unger, P.W., 1998. Distribution and amount of soil organic C in long-term management systems in Texas. Soil Till. Res. 47, 309–321.

Potter, K.N., Torbert, H.A., Johnson, H.B., Tischler, C.R., 1999. Carbon storage after long-term grass establishment on degraded soils. Soil Sci. 164, 718–725.

Potter, K.N., Daniel, J.A., Altom, W., Torbert, H.A., 2001. Stocking rate effect on soil carbon and nitrogen in degraded soil. J. Soil Water Conserv. 56, 233–236.

Pratt, P.F., Goulben, B., Harding, R.B., 1957. Changes in organic carbon and nitrogen in an irrigated soil during 28 years of differential fertilization. Soil Sci. Soc. Proc. 21, 215–219.

Ramage, C.S., 1971. Monsoon Meteorology. Academic Press, New York.

Redeker, K.R., Wang, N.-Y., Low, J.C., McMillan, A., Tyler, S.C., Cicerone, R.J., 2000. Emissions of methyl halides and methane from rice paddies. Science 290, 966–969.

Reheis, M.C., Goodmacher, J.C., Harden, J.W., McFadden, L.D., Rockwell, T.K., Shroba, R.R., Sowers, J.M., Taylor, E.M., 1995. Quaternary soils and dust deposition in southern Nevada and California. Geol. Soc. Am. Bull. 107, 1003–1022.

Reicosky, D.C., Dugas, W.A., Torbert, H.A., 1997. Tillage-induced soil carbon dioxide loss from different cropping systems. Soil Till. Res. 41, 105–118.

Rojstaczer, S., Deverel, S.J., 1993. Time dependence in atmospheric carbon inputs from drainage of organic soils. Geophys. Res. Lett. 20, 1383–1386.

Ryden, J.C., Lund, L.J., 1980. Nature and extent of directly measured denitrification losses from some irrigated vegetable crop production units. Soil Sci. Soc. Am. J. 44, 505–511.

Sahrawat, K.L., 2003. Importance of inorganic carbon in sequestering carbon in soils of the dry regions. Curr. Sci. 84, 864–865.

Salinas-Garcia, J.R., Hons, F.M., Matocha, J.E., Zuberer, D.A., 1997. Soil carbon and nitrogen dynamics as affected by long-term tillage and nitrogen fertilization. Biol. Fert. Soils 25, 182–188.

Sass, R.L., Fisher, F.M., Lewis, S.T., Jund, M.F., Turner, F.T., 1994. Methane emissions from rice fields: effect of soil properties. Glob. Biogeochem. Cycl. 8 (2), 135–140.

Schleser, G.H., 1982. The response of CO2 evolution from soils to global temperature changes. Zeitschrift fur Naturforschung 37a, 287–291.

Schlesinger, W.H., 1985. The formation of caliche in soils of the Mojave Desert, California. Geochim. Como Acta 49, 57–66.

Schlesinger, W.H., 1991. Biogeochemistry: An Analysis of Global Change. Academic Press, San Diego, CA.

Schlesinger, W.H., 1997. Biochemistry: An Analysis of Global Change, 2nd ed. Academic Press, New York, p. 588.

Schuman, G.E., Janzen, H.H., Jerrick, J.E., 2002. Soil carbon dynamics and potential carbon sequestration by rangelands. Environ. Pollut. 116, 391–396.

Sellers, W.D., Hill, R.H., 1974. Arizona Climate 1931–1972, 2nd Revised Edition. University of Arizona Press, Tucson, 616 pp.

Sheppard, P.R., Comrie, A.C., Packin, G.D., Angersbach, K., Huges, M.K., 2002. The climate of the US southwest. Clim. Res. 21, 219–238.

Staben, M.L., Bezdicek, D.F., Smith, J.L., Fauci, M.F., 1997. Assessment of soil quality in conservation reserve program and in wheat–fallow soils. Soil Sci. Soc. Am. J. 61, 124–130.

Striegl, R.G., McConnaughey, T.A., Thorstenson, D.C., Weeks, E.P., Woodward, J.C., 1992. Consumption of atmospheric methane by desert soils. Nature 357, 145–147.

Suyker, A.E., Verma, S.B., Burba, G.G., 2003. Interannual variability in net CO_2 exchange of a native tallgrass prairie. Glob. Change Biol. 9, 255–265.

Svejcar, T., Sheley, R., 2001. Nitrogen dynamics in perennial- and annual-dominated arid rangeland. J. Arid Environ. 47, 33–46.

Teague, W.R., Foy, J.K., Cross, B.T., Dowhower, S.L., 2000. Soil carbon and nitrogen changes following root-plowing of rangeland. J. Range Manage. 52, 666–670.

Tiedemann, A.R., Klemmedson, J.O., 1973. Effect of mesquite on physical and chemical properties of the soil. J. Range Manage. 28, 27–29.

Tiedemann, A.R., Klemmedson, J.O., 1986. Long-term effects of mesquite removal on soil characteristics. I. Nutrients and bulk density. Soil Sci. Soc. Am. J. 50, 472–475.

Townsend, A.R., Vitousek, P.M., Holland, E.A., 1992. Tropical soils could dominate the short-term carbon cycle feedbacks to increased global temperature. Clim. Change 22, 293–303.

Undersander, D.L., Reiger, C., 1985. Effect of wheat residue management on continuous production of irrigated winter wheat. Agron. J. 77, 508–511.

Unger, P.W., 1991. Organic matter, nutrient and pH distribution in no- and conventional-tillage semiarid soils. Agron. J. 83, 186–189.

Unger, P.W., 1995. Organic matter and water-stable aggregate distribution in ridge-tilled surface soil. Soil Sci. Soc. Am. J. 59, 1141–1145.

Unger, P.W., Baumhardt, R.L., 1999. Factors related to dryland grain sorghum yield increases: 1939 through 1997. Agron. J. 91, 870–875.

United States Census Bureau, Census 2000. http://www.census.gov/main/www/cen2000.html.

USDA, 1936. The Western Range. In: Proceedings of the 74th Congress, Senate Document 199, U.S. Govt. Printing Office, Washington, DC.

USDA-NASS, National Agricultural Statistical Service, 2000. Agricultural statistics data base. U.S. and state data: grains (verified April 2, 2004). USDA-NASS, Washington, DC. http://www.nass.usda.gov:81/ipedb/.

van Groenigen, J-W., van Kissel, C., 2002. Salinity-induced patterns of natural abundance carbon-13 and nitrogen-15 in plant and soil. Soil Sci. Soc. Am. J. 66, 489–498.

Venterea, R.T., Rolston, D.E., 2000. Mechanisms and kinetics of nitric and nitrous oxide production during nitrification in agricultural soil. Glob. Change Biol. 6, 303–316.

Virginia, R.A., Jarrell, W.M., 1983. Soil properties in a mesquite-dominated Sonoran desert ecosystem. Soil Sci. Soc. Am. J. 47, 138–144.

Virginia, R.A., Jarrell, W.M., Franco-Vizcaino, E., 1982. Direct measurement of denitrification in a *Prosopis* (mesquite) dominated Sonoran Desert ecosystem. Oecolgia 53, 120–122.

Wagoner, J.J., 1952. History of the cattle industry in southern Arizona, 1540–1940. Social Sci. Bull. 20–23 (2), 4–132.

Wang, Y., Amundson, R., Trumbore, S., 1999. The impact of land use change on C turnover in soils. Glob. Biogeochem. Cycl. 13, 47–57.

Whalen, J.K., Bottonley, P.J., Myrold, D.D., 2000. Carbon and nitrogen mineralization from light- and heavy-fraction additions to soil. Soil Biol. Biochem. 32, 1345–1352.

Wong, A., 2003. Comparative emission of methane from different rice straw management practices in California—a statewide perspective. J. Sustain. Agric. 22, 79–91.

Wright, W.E., Long, A., Comrie, A.C., Leavitt, S.W., Cavazos, T., Eastoe, C., 2001. Monsoonal moisture sources revealed using temperature, precipitation, and precipitation stable isotope time-series. Geophys. Res. Lett. 28, 787–790.

Xu, M., Qi, Y., 2001. Soil-surface CO_2 efflux and its spatial and temporal variations in a young ponderosa pine plantation in northern California. Glob. Change Biol. 7, 667–677.

Zibilske, L.M., Bradford, J.M., Smart, J.R., 2002. Conservation tillage induced changes in organic carbon, total nitrogen and available phosphorus in a semiarid alkaline subtropical soil. Soil Till. Res. 66, 153–163.