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Use of local greenhouse gas inventories to prioritise opportunities for climate action planning and voluntary mitigation by agricultural stakeholders in California

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To meet the mitigation targets set by California's Global Warming Solutions Act there is a need for locally adapted greenhouse gas (GHG) inventory methods and policy principles that help rural communities prioritize opportunities for agricultural GHG mitigation. Here, inventory methods prescribed by the Intergovernmental Panel on Climate Change and local activity data on agricultural land uses and inputs were used to conduct an inventory of agricultural emissions for a rural county in California for 1990 and 2008. Total emissions from agriculture in Yolo County were found to decline by 10.4% during this period, due to a reduction in irrigated cropland acreage, a shift towards crops which require less N, and a reduction in N rate for some crops. Average emissions per hectare of urban land were >70 times more than our estimate for irrigated cropland. This suggests that policies which protect farmland and encourage 'smart growth' may help curb future emissions. Opportunities also exist to reduce emissions through voluntary, incentive-based, and market-driven initiatives which promote the adoption of innovative agricultural practices. To be effective, local policy makers must work closely with agricultural stakeholders to anticipate and adapt to the practical tradeoffs and co-benefits of new climate policies.

Keywords: climate change; greenhouse gas inventory; agricultural emissions; voluntary mitigation; farmland preservation; avoided conversion

1. Introduction

With the passage of the Global Warming Solutions Act in 2006 (AB32), California has rapidly become the nation's foremost laboratory for climate change policy. Efforts to curb greenhouse gas (GHG) emissions in California are also indicative of a larger trend where, in the absence of cohesive federal leadership, many state and local governments have chosen to adopt a bottom-up approach to greenhouse gas mitigation (Victor *et al*. 2005, Lutsey and Sperling 2008). Specific targets set by AB32 aim to reduce California's GHG emissions to 1990 levels by 2020, and a further 80%

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by 2050. Recognising the key role that land-use planning will play in achieving these goals, legislators also passed Senate Bill 375 (SB375) in 2008, which requires regional administrative bodies to develop sustainable land-use plans that are aligned with AB32 (Hettinger 2011). Given the importance of agriculture as an economic land use, how these new climate policies intersect with agriculture will merit careful consideration by state and local policy makers.

Agriculture currently occupies 25.4% of California's total land area and generates approximately 6% of the state's total GHG emissions (NASS 2007, CARB 2009a). By contrast, urban areas in California make up only 4.9% of the land area but are the primary source of the state's transportation and electricity emissions, estimated at 39% and 25%, respectively (de la Rue du Can *et al.* 2008, Hanak *et al.* 2011, CARB 2010). Moreover, rapid urbanisation in California has contributed to the loss of nearly 3.4 million acres of farmland over the last decade and has further increased the emissions associated with urban sprawl (Liu *et al.* 2003, Norman *et al.* 2006, NASS 2007). At present, AB32 does not require agricultural producers to report their emissions, or to implement mandatory mitigation measures, as it does for California's industrial sector (CARB 2008b, Niemeier and Rowan 2009). The state is, however, encouraging farmers to institute voluntary mitigation strategies through various public and private incentive programmes (CARB 2008b). Likewise, voluntary mitigation projects within California's agriculture and forestry sectors may be permitted to sell offset credits in a carbon market which has been proposed in the scoping plan laid out by the California Air Resources Board (CARB) (Niemeier and Rowan 2009).

While the California Air Resources Board and other state agencies have taken the lead in defining these policies, much of the responsibility for climate change planning and policy implementation has been delegated to local governments. For example, AB32 and SB375 now require local governments to address greenhouse gas mitigation in the environmental impact report that accompanies any update to their general plan, or to carry out a specific 'climate action plan' (CAGO 2009). Consequently, conducting an inventory of GHG emissions is now among the first steps being taken by local governments in California as they plan for future development.

To help local governments improve the quality and consistency of their emissions inventories, state agencies have collaborated with several organisations to develop tools to standardise inventory methods. For example, the International Council on Local Environmental Initiatives has developed a software package known as the Clean Air Climate Protection Model to better align local methods with national and international standards (Kates *et al.* 1998, Ramaswami *et al.* 2008). Such inventory tools are ideal for appraising emissions from government or municipal operations, but are less useful for 'community-wide' assessments. In particular, the emissions from agriculture are often missing from existing inventory tools geared to local planners. There are several reasons for the omission of agricultural emissions; these include problems of complexity, data availability, boundary effects and consistency with methods designed for larger spatial scales (Ramaswami *et al.* 2008). Methods to estimate emissions from agriculture within a local inventory framework would be a valuable asset for those developing mitigation and adaptation strategies in rural communities.

In this paper, a local inventory of agricultural GHG emissions in 1990 and 2008 is presented for Yolo County, California. Yolo County was chosen because it has many attributes typical of the Central Valley: small towns and cities with a changing mixture of urban, suburban and farming-based livelihoods. Its agricultural landscape includes a mix of irrigated row crops and orchards grown on alluvial plains, and grazed rangelands in the uplands along the eastern edge of California's Coastal Range. The local government of Yolo County is also among the first in California to specifically address climate change mitigation and adaptation in their recently passed climate action plan (Yolo County 2010b). The mitigation and adaptation initiatives now being implemented by the local government in Yolo County thus provide the policy context for this analysis. The main objectives of this GHG inventory and policy analysis will therefore be to: (1) show how GHG mitigation can be facilitated by efforts to preserve farmland and curb urban growth; (2) examine the benefits and trade-offs of on-farm practices to reduce agricultural emissions; and (3) discuss how involving agricultural stakeholders in the planning process can strengthen mitigation efforts and lay the groundwork for future adaptation.

2. Materials and methods

2.1. Inventory methods and data sources

In this study, an inventory of Yolo County's agricultural GHG emissions is conducted for both the AB32 base year (1990) and the present period (2008). To address the wide range in data availability and analytical capacity that exists across different national or regional scales, the Intergovernmental Panel on Climate Change (IPCC) advocates a three-tiered approach (IPCC 2006). This tiered system refers to the complexity and geographic specificity of the inventory method in question, with lower tiered methods using a simplified default approach and relatively coarse activity data, while higher tiered methods involve more sophisticated models and higher resolution activity data (IPCC 2006). Higher tiered models generally produce more precise estimates of emissions and the inherent uncertainty, but their complexity and sophistication can also pose problems for local governments who often lack the financial resources, analytical capacity or activity data to carry out such analyses. In such cases, standard tier 1 methods which are tailored to locally available activity data may provide a more cost-effective, transparent and user-friendly alternative for rural communities.

The tier 1 methods used in this county-level analysis are adapted from three main sources: (1) the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC 2006); (2) the US EPA Emissions Inventory Improvement Program Guidelines (USEPA 2004, 2010); and (3) the CARB Technical Support Document for the 1990–2004 California Greenhouse Gas Emissions Inventory (CARB 2009b). Table 1 and the online supplementary material provide detailed information on local activity data, standardised equations and default emissions factors used to estimate emissions for the nine agricultural source categories examined in this study (see S1). These source categories include: direct N_2O from N applied to, or deposited by livestock on, croplands and rangelands; indirect N_2O from volatilisation and leaching; CO_2, CH_4 and N_2O from mobile farm equipment and irrigation pumping; CH_4 from livestock enteric fermentation; CH_4 and N_2O and manure management; CH_4 from rice cultivation; CO_2, CH_4 and N_2O from residue burning; and CO_2 from both lime and urea application (Table 2).

While the supplementary materials (online only) give a comprehensive summary of the inventory methods and activity data, a brief discussion of the main data types is provided here to identify possible sources of uncertainty in the analysis. Estimated

Table 1. Summary of data sources used in the inventory of Yolo County agricultural GHG emissions.

Source	Data types	Description of reference
Yolo County Agriculture Commissioner's Office (YCAC)	Harvested area by crop (ha) Crop production by crop (t) Livestock numbers by group (h) *excluding dairy cattle	1990 and 2008 Annual Crop Reports
US Department of Agriculture (USDA)	Dairy cattle numbers (h)	National Agriculture Statistical Service Online Database
University of California Cooperative Extension (UCCE)	Synthetic N application rate (kg N ha^{-1}) Diesel fuel use (L ha^{-1})	Archived Cost and Return Studies (various years)
California Air Resources Board (CARB)	Number of diesel irrigation pumps and activity data	2006 Survey of Irrigation Pump Engines
	Emission factor for rice production (kg CH_4 ha^{-1}) Fraction of crop acreage burned (%) Emission factors for residue burning (by crop)	2009 Inventory Technical Support Document
California Department of Agriculture (CDFA)	County lime and urea sales (t)	1990 and 2008 Fertiliser Materials Tonnage Reports
International Panel on Climate Change (IPCC)	Default Emissions Factors	2006 IPCC Guidelines for National Greenhouse Gas Inventories

emissions are based primarily on historical records of crop acreage and production input rates for 16 crop categories, and the population of six livestock groups (YCAC 1990, 2008, NASS 1990, 2008). Together these account for more than 90% of the county's irrigated cropland and livestock population, with the fraction remaining considered a minor source of uncertainty. Production input rates for N fertiliser and diesel fuel for the relevant time frames and crop categories are based on archived cost and return studies publicly available from the University of California Cooperative Extension (UCCE various years). While these cost and return studies represent a reasonable approximation of the input rates used by farmers adhering to recommended practices for a given crop, they do not capture the full range of technology and management practices that might affect fertiliser and fuel use. This caveat having been stated, two pieces of additional data suggest that this approach yields reasonable estimates of N fertiliser and diesel fuel use in the aggregate. First, the countywide sum of N fertiliser applied across all crops is only 10–15% more than the total amount of N fertiliser sold countywide during both 1990 and 2008 (CDFA 1990, 2008). Second, our tier 1 calculations of annual fuel use and emissions from mobile farm equipment are within 3% of countywide estimates made using the tier 3 OFFROAD model developed by CARB (Yolo County 2010a). The OFFROAD model is based on detailed state records of local equipment populations, annual usage and specialised emissions factors and thus represents a robust independent validation of our tier 1 approach (CARB 2007). Uncertainties associated with the numerous emission factors used in this study are available in the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC 2006).

Table 2. Summary of Yolo County agricultural CO_2, N_2O and CH_4 emissions (kt CO_2e) for 1990 and 2008, by source category. Estimates were made using tier 1 methods, activity data based on local agricultural practices, and default emission factors. For detailed methods see supplementary material (online only).

Source category	1990 Emissions					2008 Emissions					Change since 1990
	CO_2	N_2O	CH_4	Total	Annual	CO_2	N_2O	CH_4	Total	Annual	
	kt CO_2e				%	kt CO_2e				%	%
Direct N_2O from soil	–	126.55	–	126.55	37.0	–	97.27	–	97.27	31.8	−23.1
Indirect N_2O	–	36.43	–	36.43	10.7	–	26.68	–	26.68	8.7	−26.8
Mobile farm equipment	71.00	0.57	0.21	71.78	21.0	69.43	0.55	0.21	70.19	23.0	−2.2
Irrigation pumping	39.16	0.31	0.12	39.59	11.7	40.54	0.32	0.12	40.98	13.5	3.5
Livestock[a]	–	*10.64*	26.53	26.53	7.8	–	*12.39*	31.84	31.84	10.5	20.0
Rice cultivation	–	–	25.92	25.92	7.7	–	–	31.16	31.16	10.2	20.2
Residue burning[b]	–	4.86	1.76	6.61	2.0	–	1.59	0.83	2.42	0.8	−63.4
Lime	4.35	–	–	4.35	1.3	2.32	–	–	2.32	0.8	−46.7
Urea	4.15	–	–	4.15	1.2	3.46	–	–	3.46	1.1	−16.7
Total	118.66	168.71	54.54	341.92		115.74	126.41	64.16	306.31		−10.4

Note: [a]N_2O from N excreted by livestock (in italics) is assumed to be applied to soil as manure or urine, thus it is only included in the totals for direct and indirect N_2O.
[b]CO_2 emissions from residue burning (104.92 kt in 1990 and 42.69 kt in 2008) is considered a biogenic emission, thus was not included in the total.

3. Results

3.1. Inventory of agricultural emissions in 1990 and 2008

In Yolo County, total agricultural emissions declined by 10.4% between 1990 and 2008 (Table 2). The primary reason for this generalised decline is a notable reduction in both direct and indirect N_2O emissions (Table 2). While direct N_2O emissions were the largest source of emissions during both inventory years, emissions from this source decreased by 23.1% over the study period due to a reduction in the amount of N fertiliser applied countywide (Figure 1). This reduction in fertiliser use is driven by two important land use trends: (1) a 6% reduction in the county's irrigated cropland (Table 3, Figure 2); and (2) a general shift away from crops that have high N rates (e.g. corn, tomatoes), coupled with an expansion in alfalfa and grape area which require less fertiliser (Table 4). The large expansion of alfalfa acreage resulted in a moderate increase in the direct N_2O emissions from crop residues (Figure 1), but this increase was not enough to offset the overall savings achieved by the displacement of corn and tomatoes. The direct N_2O emissions from urine in pasture and manure application range between 5% and 15% of the total direct emissions, and showed a small rise over the study period due to a proportional increase in livestock population. Estimates of nitrate lost through leaching and runoff accounted for approximately two-thirds of the indirect N_2O emissions countywide, with NH_3 volatilisation responsible for the remaining one-third (Figure 1). More than 90% of indirect emissions originated from synthetic N fertilisers, while urine and manure from livestock were relatively minor sources. Consequently, the notable decline in indirect N_2O emissions was also due to a decrease in the amount of synthetic N applied countywide.

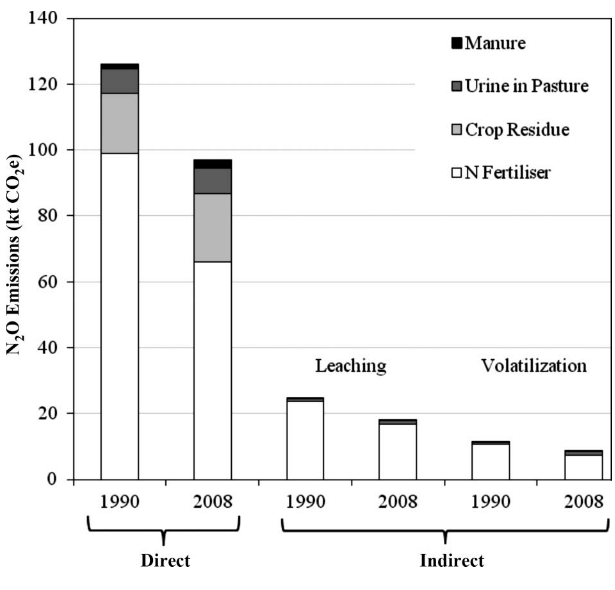

Figure 1. Direct and indirect N_2O emissions (kt CO_2e) during 1990 and 2008 as a function of N source (N fertilisers, crop residues, urine in pasture, manure), leaching and volatilisation. Emissions were estimated using tier 1 methods and activity data that reflects local crop management practices and default emission factors.

Table 3. Land area and average emissions rates (t CO_2e ha^{-1} yr^{-1}) for rangeland and irrigated cropland and in Yolo County during 1990 and 2008, estimated using tier 1 methods, activity data based on local agricultural practices, and default emission factors.

	Land area		Average emissions rate	
	1990	2008	1990	2008
Land-use category	——— ha ———		– t CO_2e ha^{-1} yr^{-1} –	
Rangeland[a]	53,419	54,946	0.70	0.80
Irrigated cropland[b]	139,407	131,439	2.19	1.99

Notes: [a]Emissions from rangeland include all emissions from livestock. [b]Emissions from irrigated cropland include emissions from all other source categories.

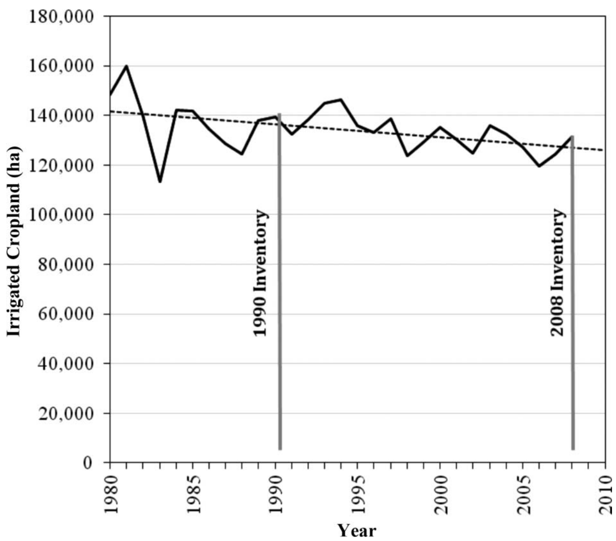

Figure 2. Change in irrigated cropland (ha) between 1980 and 2008. Vertical lines indicate when the 1990 (AB32 base-year) and 2008 inventories were conducted.

In both years, emissions of CO_2, N_2O and CH_4 from diesel-powered mobile farm equipment were responsible for 20.0 to 23.0% of total agricultural emissions in Yolo County (Table 2). While a reduction in the county's irrigated cropland might have been expected to save fuel and reduce associated emissions, this category showed little change in emissions over time (69.1 kt CO_2e in 1990 and 69.0 kt CO_2e in 2008). This was because an increase in fuel consumption per unit area for several important crops (e.g. rice, corn, tomatoes, melons and miscellaneous vegetables) offset the small decline in irrigated cropland (Table 4, Figure 2).

Diesel-powered irrigation pumps emitted approximately 39.6 kt of CO_2e in 1990 and 41.0 kt of CO_2e in 2008 (Table 2). This was equal to 11.7 to 13.5% of the total agricultural emissions. While irrigated cropland in the county decreased overall, the amount of land with access to groundwater has continued to expand as new wells are drilled. The small increase in the number of wells operating in the county accounted for the proportional rise in emissions from irrigation pumping.

Table 4. Cultivated area, production input rates and estimated emissions for crop categories in 1990 and 2008. Estimated emissions for direct N_2O, indirect N_2O, and mobile farm equipment are based on tier 1 inventory methods, local activity data and default emission factors.

			Production input rates[b]						Estimated emissions					
	Cultivated area[a]		N fertiliser		Crop residue		Agricultural fuel		Direct N_2O		Indirect N_2O		Mobile farm equipment	
	1990	2008	1990	2008	1990	2008	1990	2008	1990	2008	1990	2008	1990	2008
Crop category	ha		kg N ha^{-1} yr^{-1}				L ha^{-1} yr^{-1}		kg CO_2e ha^{-1} yr^{-1}					
Alfalfa	14,569	22,950	12	12	57	68	85	33	338	389	20	20	228	88
Almond	3054	4639	224	247	–	–	269	103	1092	1201	355	390	727	278
Corn	6070	3285	392	269	99	112	137	262	2394	1857	621	426	369	706
Grain hay	5099	6804	112	90	51	77	56	56	794	811	177	142	151	151
Grapes	640	4857	56	45	–	–	215	215	273	218	89	71	580	580
Irrigated pasture	5261	5261	50	50	–	–	2	2	246	246	80	80	6	6
Melons	2145	578	146	196	–	–	306	1169	710	955	231	310	826	3154
Prunes	880	851	168	168	–	–	168	168	819	819	266	266	454	454
Rice	10,117	12,164	191	207	12	48	186	253	337	535	302	328	502	681
Safflower	11,214	5469	112	112	–	–	122	122	546	546	177	177	328	328
Tomato	24,079	15,204	224	235	–	–	514	730	1092	1146	355	373	1387	1968
Walnuts	2739	3606	224	224	–	–	106	56	1092	1092	355	355	287	151
Wheat	28,428	17,158	224	135	68	73	115	123	1424	1008	355	213	311	333
Misc. field crops	12,100	12,309	125	125	–	–	240	227	607	607	197	197	648	613
Misc. fruit & nuts	590	619	110	140	–	–	221	190	534	682	174	222	596	512
Misc. vegetables	307	1449	232	198	–	–	816	1110	1130	966	367	314	2200	2995
Other non-specified[c]	12,115	14,236	–	–	–	–	–	–	–	–	–	–	–	–

Notes: [a]Cultivated area for all crop categories was taken from Yolo County Agricultural Commissioner's annual crop reports. [b]Inputs of synthetic N and agricultural fuel (diesel) are taken from University of California Cooperative Extension cost and return studies. [c]Inputs and emissions from the 'Other non-specified' crop category were not included in the inventory, since data on input rates were unavailable.

Emissions of CH_4 from livestock enteric fermentation and manure management contributed between 7.8 and 10.5% of the total agricultural emissions, depending on the inventory year (Table 2). This is somewhat lower than the proportion attributed to livestock statewide, which was more than 50% of all agricultural emissions in 2008 (CARB 2010). The lower figure for the county reflects the relatively small number of dairy farms operated locally. By contrast, enteric fermentation from pasture-raised beef cattle (and to a lesser degree sheep) was the largest source of CH_4 emissions from livestock in both inventory years (Figure 3). Since beef cattle and sheep populations have changed little since 1990, emissions from these livestock types were also stable over time. While dairy cattle represent only 5 to 12% of the county's cattle in any given year, an increase in the number of dairy cattle from approximately 800 to 2300 animals over the study period resulted in a 20.0% increase in total CH_4 emissions from livestock (Table 2). Emissions of CH_4 from rice cultivation increased from 25.9 to 31.2 kt CO_2e between 1990 and 2008 (Table 2). This increase in estimated emissions was due entirely to an expansion in the area under rice cultivation (Table 4).

Emissions of N_2O and CH_4 from residue burning contributed only 2.0% to the total agricultural emissions in 1990 and declined further in 2008 (Table 2). Emissions of N_2O and CH_4 were relatively small compared to the amount of CO_2 emitted during combustion (104.9 kt CO_2e in 1990 and 42.7 kt CO_2e in 2008). Most inventory guidelines consider CO_2 from residue burning to be a 'biogenic' emission, arguing that it is theoretically equivalent to the CO_2 generated during the decomposition of the same crop residue in the soil over the course of the year (CARB 2009b, IPCC 2006). Consequently, CO_2 from residue burning has been excluded from our inventory total. Emissions of CO_2 from lime and urea application each contributed approximately 1% to the overall agricultural emissions and both declined over the study period (Table 2).

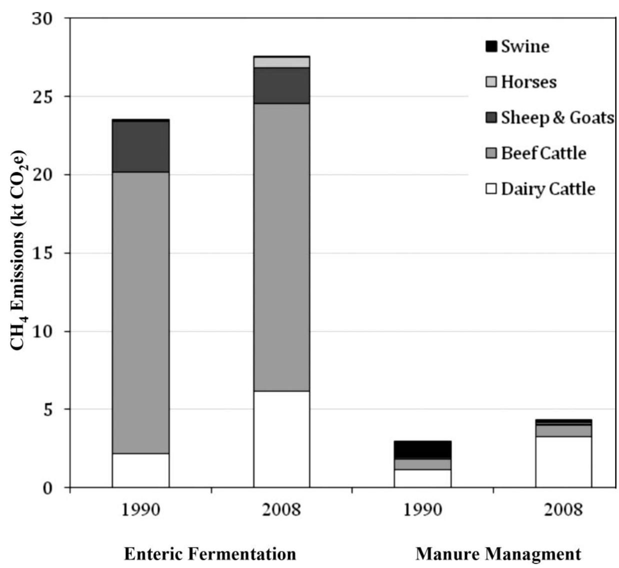

Figure 3. Livestock CH_4 emissions (kt CO_2e) from enteric fermentation and manure management as a function of livestock category. Emissions were estimated using tier 1 methods, local livestock population data and default emission factors.

4. Discussion

4.1. Changing land use patterns and its effects on emissions

One of the main findings of this study is that emissions from agriculture in Yolo County were already declining long before the implementation of recent mitigation policies. This trend is largely market-driven, arising from broad economic factors that are prompting local farmers to shift more of their land to crops which require less N fertiliser, and to adopt practices that allow inputs to be used more judiciously. As an example of the former, many local farmers point to the strong markets for wine grapes and alfalfa as the main factor behind their recent local expansion (Merenlender 2000). Other local growers surveyed during the course of this study cite the rising cost and market volatility of inputs, rather than mitigation *per se*, as a more immediate motivation to use fertiliser and other inputs more efficiently (Jackson *et al.* 2012).

Another important factor contributing to the overall reduction in agricultural emissions was the 8000 ha decline in irrigated cropland over the past two decades. This loss of irrigated cropland raises two important questions. First, what type of land-use is the cropland being displaced by? Second, how does the carbon footprint of other land-uses compare to that of agriculture? Four countywide land-use trends may explain the decline. Cropland could either be: (1) left fallow; (2) converted to non-irrigated rangeland; (3) restored to natural habitat; or (4) developed for urban and industrial use. Shifting land-use from irrigated cropland to fallow, rangeland or natural habitat will generally reduce anthropogenic emissions. The same cannot be said for cropland that is developed for urban uses. Recent studies of land-use change in Yolo County suggest that over the past two decades approximately 3000 ha of agricultural land were urbanised (CDC 2006). In 1990, urban areas (e.g. West Sacramento, Davis, Woodland and Winters) accounted for approximately 86% of the total GHG emissions countywide, while unincorporated areas supporting agriculture were responsible for the remaining 14% (Yolo County 2010a). If calculated on an area-wide basis the county's urban areas emit approximately 152.0 t CO_2e ha^{-1} yr^{-1} (Yolo County 2010a). By contrast, our inventory results indicate that irrigated cropland averaged 2.16 t CO_2e ha^{-1} yr^{-1} and that livestock in rangelands emitted only 0.70 t CO_2e ha^{-1} yr^{-1} (Table 3). This 70-fold difference in the annual rate of emissions between urbanised land and irrigated cropland suggests that land-use policies which protect existing farmland from urban development are likely to help stabilise and/or reduce future emissions, particularly if they are coupled with 'smart growth' policies that prioritise urban infill over expansion (Liu *et al.* 2003, Norman *et al.* 2006, Beardsley *et al.* 2009).

4.2. Potential for voluntary mitigation in a local agricultural policy framework

While avoided conversion of farmland may help curb emissions from urban sprawl, keeping farmland intact also affords numerous opportunities to mitigate emissions through innovative agricultural practices or by sequestering carbon in soils, perennial crops or woody vegetation (Table 5). Since N_2O emissions originating from the use of N fertilisers are the largest source of agricultural emissions, strategies to optimise N management are a high priority (Table 5). Local field and modelling studies suggest that reducing N applications, organic production and cover cropping all have potential to reduce N_2O emissions with minimal affects on crop yield

Table 5. Trade-offs and co-benefits of potential agricultural strategies to mitigate GHG emissions.

Emissions category	Strategy	Trade-offs	Co-benefits
Direct and indirect N_2O from agricultural soil	N rate reduction	– yield loss for some crops – already optimised for some crops	– lower input costs – water quality
	organic methods	– organic fertiliser costs – labour costs – limited fertiliser options – limited pest control options – yield loss for some crops	– price premium – local or direct marketing – environmental quality – agrobiodiversity
	cover cropping	– cost of crop establishment – additional fuel use – not compatible with all crop rotations – spring incorporation constraints	– soil quality – erosion and runoff control – water quality – agrobiodiversity
Mobile farm equipment	equipment maintenance	– maintenance cost – generally done already	– lower fuel costs
	optimise draw-bar load	– generally done already	– lower fuel costs
	conservation tillage	– not compatible with all crop rotations	– lower fuel costs – less labour – less wear on tractors – soil carbon sequestration – water conservation
	engine upgrades or retrofits	– cost of new equipment	– lower fuel costs – conservation of soil organic matter
Irrigation pumping	maintain pump bowl assembly	– maintenance cost – generally done already	– lower fuel or electricity costs
	solar-powered pumps	– cost of photovoltaic cell – limited to low horsepower engines – limited to daytime use	– lower fuel or electricity costs

(*continued*)

Table 5. (*Continued*).

Emissions category	Strategy	Trade-offs	Co-benefits
Livestock CH_4	biogas control systems	– cost of building the system – engines subject to air quality regs.	– energy generation (gas or electricity) – sale of carbon credits
Rice cultivation CH_4	baling and removal of straw	– baling costs – limited market for rice straw – impacts quality of waterfowl habitat	– sale of rice straw – feed and bedding for livestock – feedstock for biomass power generation
	reduce winter flooding	– poor decomposition of straw – impacts quality of waterfowl habitat	– lower pumping costs, fuel savings – water conservation
	mid-season drainage	– crop water stress – yield loss	– control of aquatic weeds – water conservation
Residue burning	minimise burning	– low overall mitigation potential – already regulated	– air quality
Urea use	substitute non urea-based N fertilisers	– low mitigation potential	–
Lime use	none proposed	– low mitigation potential	–
Carbon sequestration	reforest rangelands, riparian zones and hedgerows	– cost of establishment – access to irrigation during early years	– water quality – erosion control – biodiversity

(Krusekopf *et al.* 2002, De Gryze *et al.* 2009, 2010, Smukler *et al.* 2010). An examination of local archived cost and return studies indicates that recommended N rates have already decreased somewhat for corn, wheat, hay and grapes over the past 20 years, but have increased slightly for tomatoes, melons, rice and almonds (UCCE various years). Thus, while some growers have already improved N management, further reductions in N inputs may be possible for some crops (Cavero *et al.* 1998, Smukler *et al.* 2010). Local outreach programmes conducted in partnership with agricultural organisations and co-operative extension, which share information on practices, technologies and incentives, are needed to help growers optimise N rates while maintaining yields. Policy makers should seek opportunities to align future mitigation initiatives with these nascent efforts.

Policies which emphasise mitigation practices that offer direct private benefits are also likely to gain more traction among farmers. For example, recent studies indicate that farmers generally favour measures to conserve fuel and electricity as opposed to those that might impact yield or revenue, such as reducing N rates (Jackson *et al.* 2012). This suggests that workshops and information bulletins highlighting the possible fuel savings achieved through routine engine and pump maintenance, or more efficient field operations (e.g. optimising drawbar load, fewer tillage passes), are likely to generate more interest (Yolo County 2010b). Evidence from local interviews also suggests that incentive programmes such as California's Carl Moyer Off-Road Equipment Replacement Program, which provides financial assistance for new equipment or engine upgrades that meet or exceed state air quality standards, have already been successful in boosting farmer participation in GHG mitigation initiatives (CARB 2008a).

For livestock raised primarily on rangeland pasture, practical options to reduce CH_4 emissions from enteric fermentation and manure management are somewhat limited, because livestock managers cannot intensively manage the diet and manure of animals raised in an extensive rangeland setting. For livestock managers who run confined feeding operations, policies to help fund biogas control systems for electricity or heat generation have the potential to considerably reduce CH_4 emissions. Unfortunately, strict air quality standards which require engines that burn methane to emit less than 50 ppm of NOx can sometimes pose a disincentive for adoption in California (CARB 2001). A re-evaluation of state and local air quality regulations in light of the possible climate change benefits associated with biogas control technologies may help strike a balance between air quality and climate change objectives.

Emissions of CH_4 from rice cultivation provide another example of how differing air quality and climate change priorities can sometimes lead to policies that run contrary to one another. Prior to 1991, virtually all rice straw in California was burned in the field after harvest, a practice that led to protracted public debate about local air pollution and culminated with the passing of the Rice Straw Burning Act (Jenkins *et al.* 1992, Hill *et al.* 2006). As an alternative to burning, most rice farmers shifted their post-harvest practices to a combination of residue incorporation and winter flooding, which has led to lower yields and higher production costs (Hill *et al.* 2006). These policy-driven changes in residue and water management have improved air quality in the Sacramento Valley and enhanced winter habitat for migratory waterfowl. However, field studies testing the effects of residue incorporation and winter flooding now estimate that this policy has led to a two- to three-fold increase in the amount of CH_4 emitted from California rice fields (Bossio *et al.* 1999, Fitzgerald *et al.* 2000). Since the tier 1 approach used in this study does not include changes in cultivation practice, the possible increase in CH_4 emissions is not accounted for in our study. However, recent analysis conducted using the tier 3 Denitrification Decomposition (DNDC) model indicates that CH_4 emissions from rice cultivation countywide are likely to have increased from approximately 32 to 58 kt CO_2e over the study period, due to changes in water and residue management (Table 6). Strategies that may help reduce CH_4 emissions include baling straw for off-farm uses (e.g. bedding, energy generation, low quality feed), mid-season drainage and reduced winter flooding. However, before promoting such practices policy makers should carefully consider how they might impact grower livelihoods and the other ecosystem services provided by local rice fields.

Estimates presented in this study indicate that emissions from burning crop residues and the application of lime and urea are a very small fraction of agricultural emissions in Yolo County, and have already been declining over the past two decades. This suggests that additional policies targeting residue burning, lime and urea will have little impact on overall emissions. By contrast, recent landscape studies conducted in Yolo County suggest that programmes to sequester carbon in agricultural soils and plant biomass through various reforestation projects (e.g. in rangelands, riparian zones and hedgerows) have considerable potential to offset the county's GHG emissions (Smukler *et al.* 2010, Young-Mathews *et al.* 2010). Carbon can also be sequestered in the biomass of perennial orchard crops, however, offset protocols for these systems do not currently exist. At present, the lack of high resolution data on the diverse range of agricultural practices used in this region over the past 20 years makes it very difficult to estimate changes in soil and woody biomass carbon with any degree of accuracy. Future research could investigate how restoration efforts might be able to increase C sequestration in soil and wood using spatially-explicit modelling, with special focus on management of marginal lands. The sale of carbon offset credits is also a potential opportunity to raise funds for reforestation and farmscaping projects, assuming that future protocols to quantify and monitor local carbon storage can meet the criteria of being real, permanent, quantifiable, verifiable, enforceable and additional (Niemeier and Rowan 2009).

4.3. Choosing appropriate methods to monitor and model agricultural emissions

As local greenhouse gas inventories become more commonplace, an assortment of tools for monitoring and modelling agricultural emissions will be needed to help rural areas develop comprehensive mitigation plans for all sectors (Ramaswami *et al.* 2008). This study provides a relatively transparent, cost-effective and computationally simple set of accounting methods for a geographically-bounded agricultural GHG assessment, based primarily on guidelines set forth by the IPCC's tier 1 approach (IPCC 2006). The approach is also tailored to agricultural activity data available to many rural communities, and thus may be considered useful to local policy makers in California and elsewhere (CARB 2009b).

While the IPCC's tier 1 methods provide a reasonable approximation of emissions from various agricultural sources and thus may help local planners prioritise mitigation efforts, their computational simplicity and their use of default emissions factors have important limitations. For example, the default emissions factor used for direct N_2O emissions from N fertilisers applied to agricultural soil ($EF = 0.01$ kg $N_2O - N$ per kg N applied) is derived from a global data set of field

Table 6. Methane emissions from rice cultivation in Yolo County for 1990 and 2008 estimated using the tier 1 method and the DNDC model.

Method	Year	DNDC scenario	Harvested area ha	Residue burned % of ha	Residue incorporated % of ha	Winter flooded % of ha	Emissions factor kg CH_4 ha^{-1} yr^{-1}	Total CH_4 emissions kt CO_2e
Tier 1 Method[a]	1990	–	10,117	–	–	–	122.0	25.9
	2008	–	12,164	–	–	–	122.0	31.2
DNDC Model[b]	1990	A	10,117	100.0	0.0	0.0	151.4	32.2
	2008	B	12,164	12.5	87.5	0.0	196.0	50.0
	2008	C	12,164	12.5	87.5	100.0	257.4	65.8
	2008	Ave. B & C	12,164	12.5	87.5	50.0	226.7	57.9

Notes: [a]For the tier 1 method developed by CARB, no differences in residue and water management among growers are explicitly defined. [b]For the DNDC model, scenario A is assumed to reflect farmer practices before the 1991 passage of the Rice Straw Burning Act, while an average of scenarios B & C best approximates the range in residue and water management, as of 2008.

studies covering a wide range of climatic, soil and agronomic conditions and thus has a large uncertainty range of 0.003 – 0.03 kg N_2O – N per kg N applied (IPCC 2006). When default emission factors are subsequently applied to specific geographic regions this uncertainty can restrict the precision of tier 1 estimates (Smith *et al.* 2010). Furthermore, the IPCC's tier 1 inventory methods do not attempt to capture spatial and temporal differences in soil carbon stocks in response to changing crop management or environmental conditions.

As such, if financial resources, analytical capacity and context-specific data on crop management, emissions rates and soil carbon stocks are available, more sophisticated tier 2 and tier 3 models are generally preferred. The recently released Agriculture and Land Use (ALU) Greenhouse Gas Inventory Software is one new tool available to local governments that can accommodate both tier 1 and tier 2 inventory methods, as well as spatial data on soil carbon stocks (CSU 2012). For stakeholders and government agencies who intend to participate in emerging carbon markets, process-based biogeochemical models (e.g. DNDC, DAYCENT, COMET-VR) are typically required, as these tier 3 models generally offer more precision in their emissions estimates and greater flexibility in scenario analysis (Flynn *et al.* 2009, De Gryze *et al.* 2011). The availability of local data and region-specific models will determine the feasibility of these different types of GHG inventory methods. As scientific progress is made, we are likely to see better resolution of global vs. regional mitigation actions related to N fertiliser use, renewable sources of N inputs, and practices to increase carbon sequestration across agricultural landscapes.

5. Conclusions

As California begins to implement the mitigation policies of AB32, the present study offers several insights that will be relevant to other local governments and agricultural communities. First, since emissions from cropland and rangeland were several orders of magnitude lower than urbanised land (per unit area), local measures to protect farmland may themselves be viewed as mitigation strategies, or at the very least a means of stabilising emissions. Perhaps more importantly, the idea of 'greenhouse gas mitigation via farmland preservation' is likely to win support among rural stakeholders with long-term intentions to remain in farming, ranching and associated industries. Aligning farmland preservation policies with legislation to reduce GHG emissions (i.e. SB375 for regional planning and AB32 for reducing global warming) might also generate further backing within rural communities if it helps to justify and safeguard agriculture's unique 'voluntary' mitigation status among California's major economic sectors (Niemeier and Rowan 2009). This common ground may also help engage both stakeholders and regional planners in the broader discussion of how agriculture can adapt to the risks posed by climate change. Since farmland preservation also requires coping with climate change, more attention needs to be placed on the trade-offs that often arise between managing for mitigation of GHG emissions versus adaptation (Table 5).

While some have characterised voluntary mitigation strategies as inherently weak policy instruments (Lyon 2003), others have begun to highlight examples of how partnerships between local governments and various stakeholders can lead to substantive climate action planning and noteworthy reductions in GHG emissions (Adger 2003). In this context, bottom-up local initiatives may be more attractive to farmers in promoting more efficient use of agricultural inputs (e.g. fertiliser, fuel and water) while maintaining or enhancing crop productivity. Some initiatives by stakeholders will seek transformative change of local agro-ecolgical systems (e.g. organic agriculture), while others will choose to mitigate and adapt using an incremental and market-driven approach (Reganold *et al.* 2011). No matter what the approach, local knowledge on co-benefits and trade-offs of GHG mitigation must be shared among farmers, extension workers, researchers and policy makers in order to further empower rural communities to develop sustainable solutions (Warner 2005, Cohen and Neale 2006, Reganold *et al.* 2011).

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