Bioenergy as an Agricultural Greenhouse Gas Mitigation Strategy in Washington State

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Introduction

In addition to lowering greenhouse gas emissions and sequestering carbon, agriculture has the potential to produce energy products that can reduce emissions in the energy sector by displacing fossil fuel products. The potential greenhouse gas reductions from bioenergy could be quite significant in comparison to direct mitigation opportunities from agriculture; the IPCC (2007) estimates that biofuels have a mitigation capacity that is 20-90% of all other agricultural mitigation activities combined at a price of \$50/MT CO_2e.[1]

Agricultural bioenergy can be produced from crops grown specifically for the purpose of producing energy, or from byproducts or waste materials generated in the process of producing agricultural products for other purposes. In each of these categories, there are some existing, commercially available technologies (e.g. biodiesel production from oilseeds or compressed biogas from animal manures) while other technologies are not yet widely viable on a commercial scale (e.g. cellulosic ethanol production from switchgrass or pyrolysis of biomass).

When bioenergy is produced, emissions reductions result if the production chain for the energy (both producing / recovering the raw material and manufacturing the consumable energy product) results in lower emissions than the fossil fuel product it replaces. GHG impacts such as those shown above are generally calculated by comparing the emissions of the biofuels to emissions of the fossil fuels they replace through Life Cycle Assessment (LCA), a suite of total cost accounting methodologies for assessing the relative merit of one product over another. While a "full" LCA can cover a wide range of issues, including energy use, greenhouse gas (GHG) emissions, air pollution, soil erosion, land use, biodiversity, toxicity, water quantity, human health, and other outcomes, individual LCA studies are generally constrained by the resources and time available to the analyst. Due to these constraints, most LCAs focus on the impacts that are perceived to be most important for the given product. For biofuels, this generally means total energy, fossil fuel-derived energy, petroleum energy, and/or greenhouse gas emissions.

[1] MT = metric tons (1 MT = 1 Mg); MMT = million metric tons (1 MMT = 1 Tg)

In general, some forms of bioenergy create greater reductions in greenhouse gas emissions than others (Figure 22.1). Bioenergy produced from organic wastes tends to provide larger greenhouse gas benefits, because these products avoid the large negative GHG impacts that are associated with the production of biofuel feedstocks, and also provide a net carbon gain in cases where the waste otherwise produces large emissions (e.g see biogas from livestock manure in Figure 22.1).

Biofuels from dedicated energy crops tend to create smaller reductions in GHG emissions than biofuels from waste products because of emissions associated with crop production (e.g. see biodiesel from canola and palm oil in Figure 22.1). However, even within a single category, such as biodiesel from canola, impacts will vary depending on the details of the production process, and some portions of the lifecycle have bigger GHG impacts than others. Feedstock production generally has the greatest energy and GHG impacts, while processing has a somewhat lesser impact and transportation has a relatively small impact. Based on this assessment, purchasing decisions that encourage more energy-efficient feedstock production (including reduced fertilizer use and higher yields per unit of input) are likely to be more important in promoting substantial GHG reduction than encouraging feedstock production in Washington State or nearby areas (Kruger and Yorgey, 2008).

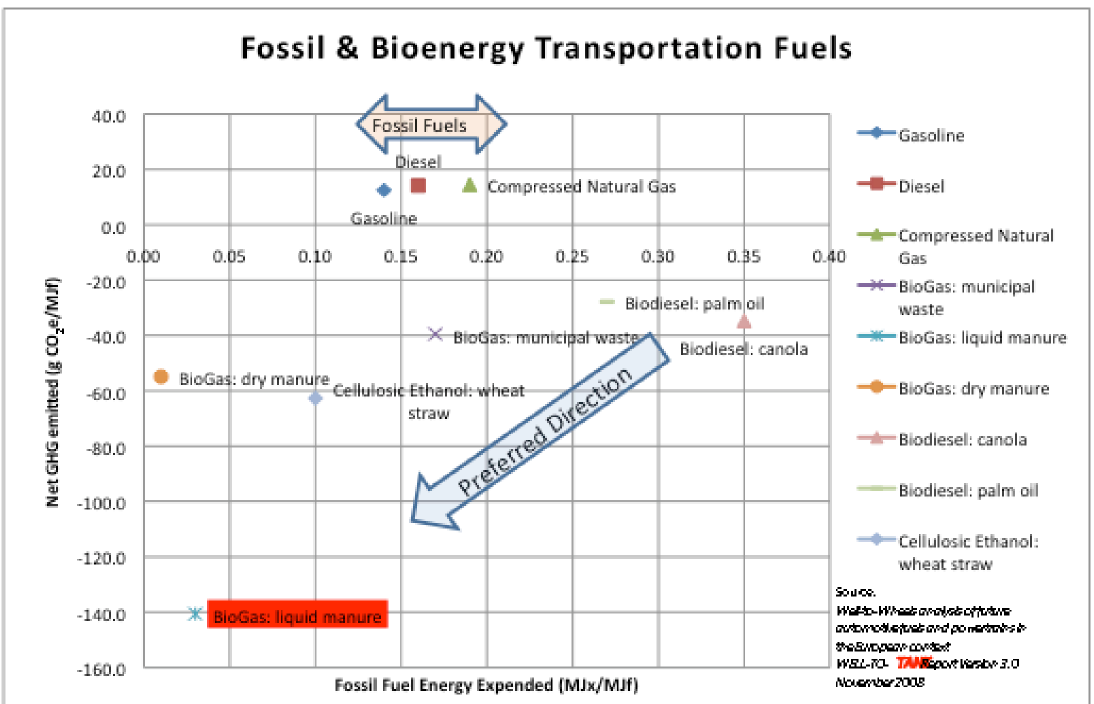

Figure 22.1. Analysis of predicted net GHG emissions and fossil fuel energy expended from wheel to wheels for a variety of future automotive fuels in the European context (data from EUCAR, et al., 2008).

In Washington State, as at the national level, there has been considerable interest in bioenergy from both crops and organic wastes. While climate change impacts are one reason for this interest, other negative environmental impacts of petroleum product use such as air quality concerns are also important. So are other non-environmental concerns such as lessening our dependence on foreign oil and improving our national security. In 2007, the United States imported 13.5 million barrels per day of imported crude oil and petroleum products, representing 58% of consumption (EIA, 2009). Bioenergy also has the potential to contribute to economic development, particularly in rural areas. The USDA (2008) estimated that 17,000 jobs are created for every billion gallons of biofuel produced.

In response to ongoing interest in bioenergy in the Pacific Northwest, and to complement the large body of existing and ongoing research into bioenergy technologies, the Climate Friendly Farming Project has focused its efforts on two specific areas that had particular merit for research and development in the state.

First, we have focused on evaluating the role that biofuels crops can play as rotational crops in our dominant cropping systems in the region. This approach is particularly relevant to the Pacific Northwest, where biofuel crops (such as oilseeds) are unlikely to ever become a primary crop choice for farmers, due to the existence of more valuable primary crops such as wheat in dryland areas, and potatoes, hay, fruits and vegetables in irrigated regions. However, biofuels crops have been shown to provide benefits to the sustainable production of the primary crop, by breaking up pest cycles (weed, insect and diseases), minimizing synthetic chemical needs for primary crops, and in some cases improving yields of the primary crop (Guy and Gareau, 1998; Guy et al., 1995). By testing varieties, running breeding programs, doing economic analysis, and providing growing information suited for the Pacific Northwest, it is possible that some bioenergy crops a viable rotational choice for farmers. These types of financial and weed- and disease-control considerations are among the many issues that are not captured within most LCAs, though this may improve in the future as crop and soil process models are integrated with LCA accounting methodologies.

Second, we have focused research on assessing biomass feedstock opportunities and trade-offs associated with next generation biofuel technologies. While producing biofuels from non-crop biomass suggests much greater levels of GHG mitigation potential than first generation biofuels on a life-cycle basis, it also raises a whole suite of new sustainability considerations, many which have not been adequately vetted by the scientific community. Even the most promising next generation biofuel technologies are not necessarily immune to the types of unintended consequences that have become a magnet for criticism in the ethanol and biodiesel industries. Clarens, et al. (2010) have even raised sustainability concerns about the 'holy grail' of biofuel technology, algal based biofuels, on the basis of the required nitrogen fertilizer inputs. By bringing attention to the opportunities and tradeoffs associated with biomass feedstock for next generation biofuels, we hope to identify the most promising strategies for minimizing tradeoffs where possible, and provide decision support tools for situations where trade-offs are inevitable.

Dedicated Biofuel Crops

Though biofuel crops have shown significant potential to contribute to rural economies, improve agricultural producers' incomes, and mitigate greenhouse gas emissions, these crops are not currently being grown in large amounts in the PNW, and there is much that is unknown about how well they will perform over the long run. In Washington, most feedstock crops have received little or no previous agronomic or varietal adaptation for the region. Therefore, our research in this area has focused on evaluating crop adaptation and productivity, and developing growing guidelines for producers for this region. The focus is on crops that can complement major cash crops in existing rotations, and that provide benefits to the entire cropping system. Specifically, we have focused on a variety of oilseeds for biodiesel production in eastern (both dryland and irrigated) and western Washington, and perennial grasses for cellulosic ethanol production in central Washington and the high-rainfall region of the Palouse.

Simultaneously, we have evaluated several potential economic and environmental tradeoffs for biofuel crop production. Economic and crop rotational tradeoffs affect whether or not farmers will integrate a biofuel feedstock crop as part of their rotation. Production of long-lived, perennial biofuel grasses will likely have impacts on carbon sequestration and soil nutrient management which may influence the overall net greenhouse gas balance resulting from their adoption.

The limited work by the Climate Friendly Farming team in this area helped to contribute toward reaching a critical mass for biofuels research in the state. Through the initiative of our group and many others working on biofuels issues throughout the state, research on biofuels is now being coordinated through the Biofuels Cropping Systems Research and Extension Project (http://css.wsu.edu/biofuels/index.html). This new, focused effort on biofuel crop research, development and commercialization enables improved coordination of state-wide research and extension education efforts to address priority biofuels issues throughout the state.

Oilseed Variety Trials in Irrigated Systems

Biodiesel includes fuels derived from soybeans, sunflower, cottonseed, canola and rapeseed, crambe, safflower, flaxseed, and mustard seed among others. Canola, rapeseed and camelina may fit well into existing dryland and irrigated crop rotations and will most likely be the principal biofuel oilseeds of the Pacific Northwest. However, these potential crops (and other oilseeds) have only received limited research attention in this region, and most canola is produced in North Dakota or the Canadian Prairie. Currently, soybeans (not traditionally grown in large amounts in the Pacific Northwest) are the most commonly used biodiesel feedstock in the U.S., whereas rapeseed is the primary feedstock in Europe.

In the spring of the 2004 growing season, Hal Collins (USDA-ARS) and a team of WSU and USDA scientists established an irrigated trial of oilseed crops on the USDA

Paterson experimental farm to evaluate the potential for biodiesel and ethanol production. Crops tested included rapeseed and canola (*Brassica napus* or *B. campestris),* mustard (*Sinapsis alba)*, crambe (*Crambe abyssinicia*), sunflower (*Helianthus*), safflower (*Carthamus tinctorius*), and soybean (*Glycine max*). Background information on the major crops, plus cultural recommendations suitable for growers in the Pacific Northwest that were developed through the project are presented in more detail in Appendix 22A and in Painter et al. (2006).

Though other oilseeds also have potential, canola production (Figure 22.2) appears to be the most ready choice for biofuel crop production for most areas with sufficient moisture in the Pacific Northwest. It grows well (once a stand is established) in most agricultural areas of the region, has a high oil content (approximately 40%, compared to about 30% for yellow mustard and 20% for soybeans), and the meal byproduct, low in erucic acid and glucosinolates has a ready local market as dairy feed. Canola and rapeseed, particularly the winter varieties, produce high quality oil suitable for biofuel production.

Figure 22.2. Canola/Rapeseed.

Results of oilseed variety trials are shown in Table 22.1. Data from the trials indicate that 50,000-80,000 acres would be needed to support a single 5 million gallon per year biodiesel facility depending upon which oilseed was grown. Yields of spring mustard averaged about 90% of their expected yield potentials. Soybean yields were greater than expected, averaging over 60 bushels acre^{-1} compared to the national average of 48 bushels acre^{-1} under irrigation. Since soybean required similar planted acreage to safflower or canola (dependent upon variety) for a similar biodiesel yield, soybean production in the Columbia Basin might have improved GHG impacts compared to other crops, as soybeans fix their own N. However, a regional market would need to be developed for soybean oil biodiesel production to be viable and Life Cycle Analyses are needed to confirm the GHG benefits of the reduced applications of N resulting from a soybean oil/biodiesel production system.

Safflower yields were 50% of expected yield, with weed control representing the greatest challenge. Crambe was the poorest performer in our variety trials due to significant weed competition and the lack of herbicides available to control broadleaf weeds. Weed problems and low economic returns also make meadow foam and camelina a high risk at this time.

To meet the requirement of the 2% biodiesel mandate passed in Washington State in 2006 approximately 20 million gallons of biodiesel would be needed. Given the yields seen in our trials, if all the oil was to come from Washington growers, 250,000-350,000 acres of dryland and irrigated crop land would be needed. Continued improvements for breeding high seed oil contents and reducing yield variability through improved crop management may greatly reduce the acreages needed.

Further reporting on oilseed variety trials will be made available on the Biofuels Cropping Systems Research & Extension Project website: http://css.wsu.edu/biofuels/index.html.

Table 22.1. Yield data from the 2004-2009 irrigated biofuel variety trials at Paterson, WA and estimates of land area needed to support a 5 million gallon per year biodiesel facility.

Crop/Variety	Yield (lb ac^{-1})	Oil (%)	Biodiesel Yield (gal ac^{-1})	Acreage to support 5 MGY facility
Crambe				
Belann	830	11.6	12.6	397,445
Meyer	1056	17.3	23.9	209,430
Spring Mustard				
Pacific Gold	2194	28.8	82.6	60,545
Soybeans				
S1918-4	3881	17.6	89.2	56,044
S2422-2	3897	18.5	94.2	53,097
S2100-2	3510	15.6	71.6	69,875
S2788	3304	17.3	80.3	62,283
87009	3645	20.6	98.1	50,961
232	3564	18.7	87.1	57,415
IA1007	3383	16.4	72.5	68,970
IA1008	3546	19.1	88.5	56,498
IA1010	2605	18.1	61.6	81,156
IA1013	3216	16.8	70.6	70,824
Spring Rapeseed				
Garnet	1876	32.7	80.2	62,364
Sterling	1770	33.8	78.1	63,997
Safflower				
Montola	1814	32.9	78.2	64,136
OW74	2015	33.1	86.9	57,546
CW990L	3250	40.0	170.0	29,412
S345	3560	40.0	186.1	26,882

Tradeoffs: Economics of Spring and Winter Canola Production in Dryland Eastern, Irrigated, and Western Washington Agricultural Systems

Because canola is grown in rotation with other crops, analysis of the economics of canola must explicitly incorporate an analysis of the entire rotation. To help growers and others understand these financial dynamics, we developed enterprise budgets for spring and winter canola based on interviews with producers in early 2006, yield data from growers and variety trials carried out in our region, and input and harvest prices gathered from public sources in 2007 (for more detailed methods, see Painter and Roe, 2007). The budgets are necessarily based on farm conditions that are generalized, rather than those of a specific farm, and on prices that are constantly changing.

The analysis was completed using cost and price data from 2007. The last several years have witnessed extreme volatility in prices for agricultural inputs and products, most dramatically for grains, which were inflated in 2007 compared to long-run averages. However, for most inputs and other agricultural products, 2007 prices were fairly close to 5 year averages and therefore did not impact the analysis greatly.

While these enterprise budgets are designed so that individual farmers can adapt them to their specific circumstances and use them to support decision-making, they are also useful tools for exploring the general economic dynamics driving producers' decisions to grow (or not grow) crops such as canola, and our focus here will be on these more general lessons.

For dryland producers in eastern Washington, spring canola returns over variable production costs (costs directly associated with producing the crop, including seed, fertilizer, fuel, machinery repairs, etc.) ranged from $33/ac to $102/ac based on 2007 prices (Table 22.2). However, these variable production costs do not account for opportunity costs such as the fact that the land could be used for a purpose other than growing canola. To incorporate this, returns to total costs includes land rent (rental values are higher where alternate uses generate higher returns), investments in machinery and other fixed costs, and the operator's labor. Returns above total production costs represent returns to management and risk. Spring canola was not profitable in the driest areas when considering total production costs (Table 22.2). However, producers were able to produce profits above total costs in areas with more than 15" of rainfall.

Table 22.2. Returns over variable and total production costs for spring canola in dryland eastern Washington.

	Price ($/cwt)	Yield (cwt/ac)	Total Cash Receipts ($/ac)	Variable Production Costs ($/ac)	Returns over Variable Production Costs ($/ac)	Total Production Costs ($/ac)	Returns over Total Production Costs ($/ac)
Eastern WA:							
>20” rainfall	15.00	16	$240	$138	$102	$188	$52
15”-20” rainfall	15.00	13	$195	$137	$58	$179	$16
12”-15” rainfall	15.00	10	$150	$117	$33	$153	-$3

In addition to thinking about total costs, it is important to compare the returns from canola with other crops that could replace canola. In dryland areas where wheat is the main crop, canola would replace current rotational crops of garbanzos, lentils, or dry peas. In a generalized dryland system, based on August 2007 prices, canola was less profitable than garbanzos or lentils (Table 22.3). Shifts in the relative prices for these alternate crops could change the outcome of this analysis.

Table 22.3. Comparison of net returns for major grain crops and common rotational crops relative to spring canola in eastern Washington under dryland production.

Major Grain Crops and Common Rotational Crops (shaded)	Yield	Revenue	Variable Production Costs	Returns over Variable Production Costs	Total Production Costs	Returns over Total Production Costs
	(unit/ac)	($/ac)	($/ac)	($/ac)	($/ac)	($/ac)
Winter Wheat	82	$533	$167	$366	$333	$200
Spring Wheat	65	$423	$202	$221	$320	$103
Spring Barley	2	$376	$149	$227	$277	$99
Dry Peas	2000	$220	$141	$79	$227	-$7
Lentils	1200	$216	$107	$109	$192	$24
Garbanzos	1200	$336	$167	$169	$289	$47
Spring Canola	1700	$255	$184	$71	$263	-$8

Price assumptions:
winter and spring wheat: $6.50/bu
barley: $188/ton
dry peas: $0.11/lb
lentils: $0.18/lb
garbanzos: $0.28/lb
spring canola: $0.15/lb

However, the analysis above does not account for possible impacts that one crop in the rotation can have on yields of another crop in the rotation. Growers frequently see a significant yield boost in wheat following peas as well as reduced need to purchase nitrogen inputs because of residual nitrogen. Thus peas are a popular rotational crop for wheat producers despite their low profitability. Canola growers also report a yield advantage in the following wheat crop, but canola does not provide the nitrogen benefit.

Similar to spring canola, an analysis of winter canola must account for the economic returns on the crops it replaces. Winter canola can be grown following summer fallow in dryland areas east of the Cascades, preferably with 11” or more of rainfall, and can also be grown under irrigation in the Columbia Basin. In both systems, winter canola replaces winter wheat, though winter wheat has different roles in the two systems (as a primary crop in dryland systems, and a rotational crop with vegetables or potatoes in irrigated systems). Production costs for winter wheat and winter canola were similar, but canola yields are more variable as the crop is more difficult to establish and less winter hardy. With August 2007 prices, net returns over total production costs for winter canola averaged $99 per acre for dryland production and $141 for irrigated production (Table 22.4).

However, given the record high prices for wheat at the time (double the prices in August 2006), net returns were considerably higher for winter wheat, at $145 per acre for dryland production and $405 per acre for irrigated production (Table 22.4). These record high prices make canola look relatively less profitable than usual. In August 2006, for example, when wheat prices were $3.50 per bushel and canola was $0.12 per lb, returns on canola were $20 per acre greater than those for wheat.

Table 22.4. Returns over variable and total production costs for winter canola and winter wheat for dryland and irrigated production (canola price = $0.15/lb, wheat price = $6.50/bu).

	Yield (unit/ac)	Revenue ($/ac)	Variable Production Costs ($/ac)	Returns over Variable Production Costs ($/ac)	Total Production Costs ($/ac)	Returns over Total Production Costs ($/ac)
Dryland Winter Canola	20	$300	$116	$184	$201	$99
Dryland Winter Wheat	50	$325	$83	$242	$180	$145
Irrigated Winter Canola	36	$540	$289	$251	$399	$141
Irrigated Winter Wheat	130	$845	$282	$563	$440	$405

Even when winter canola is less profitable than winter wheat, winter canola production can make a significant difference in the overall economics of the farm through its impact on wheat yields. Growers report that reductions in herbicide needs (grassy weeds are controlled during canola production while broadleaf weeds are controlled during the wheat year), improved disease management and other benefits have created yield gains in the following winter wheat crop – particularly for farms using conservation tillage or no-till systems.

Table 22.5 shows average returns per crop year for a four-year rotation of fallow-winter canola-fallow-winter wheat, based on one farm in Adams County in an 11" annual rainfall zone (C. Hennings, personal communication, 2006) contrasted a typical two-year rotation of winter wheat-summer fallow with a four-year rotation of fallow-winter canola-fallow-winter wheat. In the typical two-year rotation, average winter wheat yields (based on county averages) were 45 bushels per acre, with an economic return of $101 each time that wheat is produced, or $50 per year. Based on the grower's records over more than 20 years, winter wheat in his four-year rotation had a 22% yield advantage of 55 bushels per acre. If a conservative canola crop failure rate of 20% is assumed due to poor conditions at seeding, average returns were $77 per acre per year, 54% higher than the two-year rotation returns.

Table 22.5. Returns over variable and total production costs for winter canola-fallow-winter wheat-fallow rotation for 10”-11” rainfall zone.

	Price ($/cwt)	Yield (cwt/ac)	Revenue ($/ac)	Variable Production Costs ($/ac)	Returns over Variable Production Costs ($/ac)	Total Production Costs ($/ac)	Returns over Total Production Costs ($/ac)
Summer Fallow							
Winter Canola	$15.00	20	$300	$116	$184	$201	$99
Summer Fallow							
Winter Wheat	$6.50	55	$358	$83	$275	$180	$178

In western Washington, the climate is also promising for rotational canola production. Field trials of spring canola in Snohomish County in 2006 yielded 2600-3000 lbs/acre, depending on the variety. Winter canola may also be well suited to this climate if disease issues associated with overwintering can be overcome, as similar climates in Germany, Belgium, and France achieve average yields of 6,000 lbs/acre. In addition, canola meal, a byproduct of biodiesel production, is in high demand for use as an animal feed in western Washington, which imports the majority of its protein requirements for animal feed.

Our economic analysis of spring canola production in western Washington showed relatively high variable production costs, due to higher custom rates and intensive tillage practices used on the heavy, high organic matter soils. Returns over total production costs (which include fixed costs such as land rent and machinery depreciation) for spring canola were $23 per acre (Table 22.6).

Table 22.6. Returns over variable and total production costs for spring canola production in western Washington, 40” rainfall.

	Price ($/cwt)	Yield (cwt/ac)	Total Cash Receipts ($/ac)	Variable Production Costs ($/ac)	Returns over Variable Production Costs ($/ac)	Total Production Costs ($/ac)	Returns over Total Production Costs ($/ac)
Western WA:							
40” rainfall	15.00	28	$420	$280	$140	$397	$23

Land rents may have been underestimated in this analysis, given the production of high-value horticultural and nursery crops in the region. Because canola cannot be grown continuously, canola would need to be incorporated into a rotation that returns adequate value, over the rotational cycle, to account for other possible uses of these rich, rain-fed soils. Two other barriers that would need to be addressed in western Washington are inadequate grain/oilseed harvesting infrastructure and potential seed contamination issues that canola production poses to the vegetable seed industry.

One final issue for growers, not reflected in the preceding analyses, is the significant year-to-year yield variability of canola. As one example, Figure 22.3 shows the yield variability of winter and spring canola on the Cook Agronomy Farm, near Pullman, WA. This creates substantial year-to-year financial risk for farmers, making it harder for them to justify planting the crop, even when average returns are acceptable. Ongoing varietal research and improved crop management practices will help develop crops and agronomic tools better suited to this region, lowering variability and making canola a more attractive crop to include in rotations.

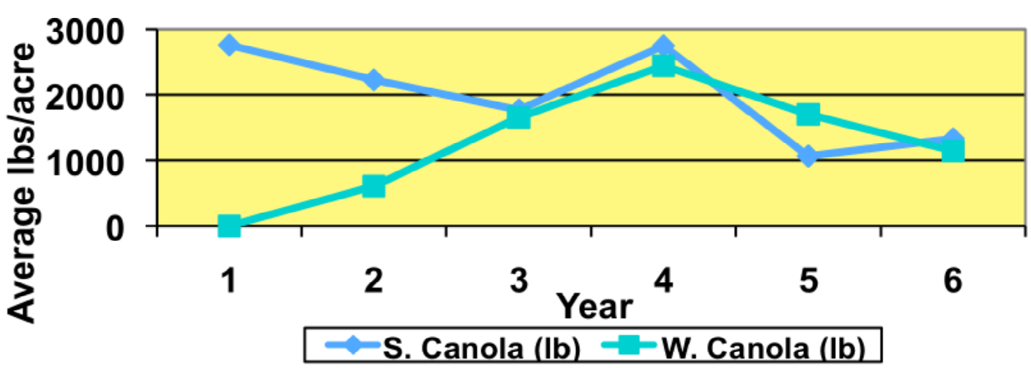

Figure 22.3. Crop Yields by Year, Cook Agronomy Farm

Switchgrass Variety Trials

Switchgrass (*Panicum virgatum*) is a warm-season, deep-rooted perennial grass species with the potential to supply biomass for cellulosic ethanol production or direct energy generation through combustion. While not native to the Pacific Northwest, the grass has been successfully produced as a seed crop in the warmer and irrigated areas of the region for more than 20 years.

As a biofuel crop, switchgrass has several potential benefits as a bioenergy crop. Switchgrass has relatively low fertilizer requirements and fewer pest issues at this time. Because it is a perennial, it does not need annual tillage. Thus soil erosion can be greatly reduced, and carbon sequestration is likely to be enhanced (see below for our initial results). Switchgrass becomes dormant and can survive under extended water stress, meaning it still produces a harvestable biomass, an important benefit over corn (which senesces and produces little harvestable yield under similar conditions), especially given that future climate change may cause rainfall to be more variable and irrigation water to be subject to regular restriction.

During the past five years we have established eight field research studies at Prosser and Paterson, WA evaluating switchgrass cultivars and production management under irrigation (Figure 22.4). Our trials focused on a number of different varieties, including Alamo, Kanlow, Dacotah, Cave-In-Rock, Trailblazer, Blackwell, Nebraska 28, Sunburst, Forestburg, and Shawnee. The most closely evaluated varieties included one lowland type, Kanlow (2n=36), and two upland types, Cave-in-Rock (2n=72) and Shawnee (2n=72). Lowland types are normally taller and coarser than upland types, and they grow more rapidly, with a more bunchgrass growth habit.

Figure 22.4. Switchgrass variety trials at Paterson, WA.

Our experience since 2004 establishing and growing switchgrass has confirmed that the crop is well suited to irrigated production in central Washington, and allowed us to develop a comprehensive set of cultural recommendations for growers, described in Appendix 22B and more fully by Fransen and Collins (2009a; 2009b). These recommendations will continue to be refined over time, and provide an important base of knowledge for growers who want to experiment with the crop.

The yields of 3-yr old switchgrass trials grown at the Paterson field site are provided in Table 22.7. Yields of the three varieties after three seasons ranged from 9 to 14 tons of dry matter per acre for two cuttings.

Table 22.7. Switchgrass biomass and ethanol yield data (Paterson, WA) and estimates of land area needed to support a 20 MGY ethanol facility.

Crop	Biomass Yield t/ac	†Ethanol Yield gal/ac	Irrigated Acres Needed to Support a 20 MGY Facility acres
Wheat Straw	5.8	£400	83, 300
Corn (grain)	5.6	580	34, 500
Corn (stover)	6.7	£464	71, 800
Corn (G+S)	11.3	1, 044	23, 300
Switchgrass			
Cave'n Rock	9.4	752	26, 600
Shawnee	10.3	824	24, 300
Kanlow	13.1	1, 048	19, 100

†Ethanol recovery from wheat straw and corn stover is estimated at 69 gallons ton^{-1}, from corn starch at 92 gallons ton^{-1} and switchgrass biomass at 80 gallons ton^{-1}.
£Assumes 60% removal of residues. €Acreage based on percentage of current forage and hay cropland.

Of the three varieties grown Kanlow is the most promising cultivar for production in the south Columbia Basin. Conservative estimates of ethanol yield ranged from 752-1,048 gallons per acre with an estimate of 20-30,000 acres needed to support a 20 MGY ethanol facility. For comparison, wheat straw and corn stover residues would need to be collected from over 70,000 acres to support the same facility, assuming 60% of the residues were harvested. Determination of ethanol production through laboratory analysis is needed to verify these estimates.

Though our production experience is still limited compared to other areas of the U.S. where switchgrass is native, we have identified two important results. First, beginning with the initial planting in 2002, we have seen that yields continue to increase each year as stands mature, and the crop is managed for biofuel. Second, a comparison of yields of Kanlow and Cave-In-Rock in several states show the yield potential of switchgrass production in Washington is on par with states where it is native (Table 22.8). In fact, second year production yields in Washington are similar to those reported in the Midwest with six-year old stands (switchgrass develops mature plants within three to five years). The high yields we have recorded on juvenile stands suggest mature stand yields could be greater than those recorded where switchgrass was first adapted, though continued trials will be needed to confirm this.

Table 22.8. Switchgrass yields of several states across the U.S.

State	Switchgrass variety	
	Kanlow	Cave In rock
	----- T ac^{-1} -----	
Texas	4.5	2.4
Upper South	5.5	4.2
Alabama	8.3	4.2
Iowa	5.8	--
Nebraska	9.2	7.3
Washington	**14.8**	**9.4**

Values presented are the sum of two cuttings per year.

In addition, an economic analysis is needed. Hay growers have equipment suitable for switchgrass and may be good candidates for growing it. Profitability will be largely determined by the transport distance and cost to the processing facility. Economic analyses should address how the crop will be incorporated into existing rotations.

One barrier to biofuels production may be lack of pesticide and herbicide registrations for these crops. No herbicides are currently labeled in the state of Washington for switchgrass planted for biofuel production. To help support future efforts, researchers tested and identified pre- and post-emergence herbicides that control most annual weeds with little injury to switchgrass (R. Boydston, unpublished data, 2010).

Possible Tradeoffs: Soil Carbon Dynamics under Switchgrass Production

While switchgrass production may mitigate GHG impacts from burning fossil fuels, there are few assessments of how removal of switchgrass biomass for fuel production will impact C dynamics within the soil. Producers and policy makers need to more fully understand the impacts of switchgrass production and removal on soil C and nutrients to help assess the long-term sustainability of biomass production. The amount of soil organic C is a function of the rates of C gains and losses from the soil under a specific land use as well as the quantity and quality of organic matter inputs (Paustian et al., 1997; Paul et al., 2001; Johnson et al., 2006). With the removal of large amounts of aboveground biomass during twice-yearly harvests, soil C levels could be adversely affected.

But switchgrass production may offset these losses. Perennial cropping conserves carbon that would otherwise be lost during annual tillage operations; perennial bioenergy crops have been shown to improve soil quality, enhance nutrient cycling, improve wildlife habitat and sequester C (McLaughlin and Kszos, 2005; Lemus and Lal, 2005). And switchgrass has a prolific root system that adds significant quantities of organic matter to the soil as roots slough and decay (Garten and Wullschleger, 2000; Ma et al., 2000a; Liebig et al., 2005; Liebig et al., 2008).

In addition, the site history of areas dedicated to perennial grass production also matters. In the Columbia Basin the use of irrigation and intensive cropping with organic additions has resulted in increases of soil organic C in rotations dominated by small grains, forages and pastures across a range of soil types (Cochran et al., 2007; WSDOE, 1999). These increases in soil organic carbon are principally due to the increase in net primary productivity and greater incorporation of residue-C into soil organic matter compared to the native shrub-steppe soils.

To better understand the carbon dynamics, we characterized the soil carbon inventories under switchgrass biomass crops, clarified the contribution of switchgrass belowground biomass production to soil organic carbon, and determined the mean residence time (MRT) of the native soil carbon and the newly incorporated carbon after three years of planting to switchgrass.

Aboveground and root biomass and C content were determined through sampling, and carbon mineralization rates were determined using the static-incubation method (see Collins et al., *in press* for a full description of methods). The CO_2 evolved during C mineralization was used to estimate the size and turnover rates for each of three carbon pools: an active pool (C_a) consisting of labile C with a mean residence time (MRT= 1/k) of days; an intermediate or slow pool (C_s) consisting of structural plant residues and physically stabilized C with an MRT of 25-50 yr; and a resistant fraction (C_r) consisting of chemically stabilized C with an MRT of 1000-1500 yr (Paul et al., 2006; Paul et al., 2001; Collins et al., 1999; 2000; Buyanovsky et al., 1994).

For soil C, it was possible for us to identify the carbon coming from switchgrass versus that already present in the soil because switchgrass is a C_4 plant that leaves a "fingerprint" of a distinctive carbon isotope (^{13}C) in the soil when its tissues decompose. Thus, if the previous cropping history was from non-C_4 crops, it is possible to determine the source of soil C, turnover rates and C sequestration (Qian and Doran, 1996; Gregorich et al., 1996; Collins et al., 1999; Clay et al, 2007). For our trials, switchgrass was planted in 2004 on soils that had been native shrub steppe until 2002, and then cropped to field corn (*Zea mays*) in 2002 and potato (*Solanum tuberosum*) in 2003. [2] Data on initial soil properties are available in Appendix 22C.

Biomass yields of the three switchgrass cultivars (Kanlow, Shawnee, and Cave-in-Rock) for three years of production are shown in Table 22.9. In the first establishment year, much less biomass was produced; in 2005 and 2006, aboveground biomass production averaged 21.8, 16.8 and 15.3 Mg dry matter ha^{-1} for the Kanlow, Shawnee, and Cave in Rock cultivars, respectively. These yields, produced under irrigation, were at the upper end of biomass production reported for 3 yr old switchgrass stands elsewhere in the United States. Across the Midwest and Southeastern U.S., annual dry matter biomass production for switchgrass ranges

[2] Field corn also uses the C_4 pathway to incorporate carbon, so contributions from the field corn were estimated using soil samples collected prior to switchgrass establishment.

from 5 - 25 Mg ha^{-1} depending on cultivar, soils and climate under rain-fed conditions (Sanderson et al., 1996; Zan et al., 2001; Lemus et al., 2002; McLaughlin and Kszos, 2005).

As expected, a sizable amount of carbon was removed each year from switchgrass harvest. Average annual C removed in the biomass for all cultivars was 8 Mg C ha^{-1} for the second and third years of production (Table 22.9).

Table 22.9. Aboveground biomass, C and N of three switchgrass cultivars.

Variety	Aboveground Biomass	C	N	C:N
2004	Mg ha^{-1}	---------kg ha^{-1} ---------		
Kanlow	3.3 a	1 360 a	26 a	52.3 a
Shawnee	2.9 a	1 145 a	18 a	63.6 b
Cave in Rock	3.0 a	1 352 a	18 a	75.1 c
2005				
Kanlow	21.0 a	9 064 a	297 a	30.5 a
Shawnee	15.1 b	6 580 b	211 b	31.2 a
Cave in Rock	13.9 b	5 600 b	180 b	31.1 a
2006				
Kanlow	22.6 a	9 754 a	275 a	35.5 a
Shawnee	18.4 ab	8 412 b	241 ab	34.9 a
Cave in Rock	16.7 b	7 582 b	216 b	35.1 a
3 yr Cumulative				
Kanlow	46.9 a	20 178 a	598 a	33.7 a
Shawnee	36.4 b	16 137 b	470 b	34.3 a
Cave in Rock	33.6 b	14 534 c	414 b	35.1 a
Average	39.0	16 950	494	36.7

DM – dry matter. Values within a column within a year within a year followed by the same letter are not significantly different at $P \leq 0.05$.

As shown in the data in Appendix 22C, root biomass was substantial, and was concentrated in the surface 30 cm of soil. Root biomass in the surface 30 cm of soil accounted for 68%, 80% and 74% of the total root biomass for the Kanlow, Shawnee and Cave in Rock cultivars, respectively. The greater root density in the surface 30cm was typical for switchgrass as reported by Ma et al. (2000a) and others (Zan et al., 2001; McLaughlin and Walsh, 1998). Soil profile root biomass produced in 2006 after three seasons of growth contained 3.3, 3.9 and 4.4 Mg C ha^{-1} m^{-1} for the Kanlow, Shawnee and Cave in Rock cultivars, respectively (Table 22.10).

Table 22.10. Carbon contained in roots of three year old switchgrass stands.

Depth	Root Carbon					
	Kanlow		Shawnee		Cave in Rock	
cm	g kg^{-1}	kg ha^{-1}	g kg^{-1}	kg ha^{-1}	g kg^{-1}	kg ha^{-1}
0-15	421 a	1546 a A	432 a	1 827 a A	443 a	2 119 a AB
15-30	408 b	756 b A	434 a	1 353 a B	426 b	1 377 b B
30-45	384 b	644 b	395 b	543 b	408 b	657 c
45-60	339 c	246 c	320 c	103 c	291 c	190 d
60-75	321 cd	80 d	269 d	24 c	243 cd	71 d
75-90	311 d	41 d	293 cd	5 c	232 d	23 d
Total		**3 313** A		**3855** AB		**4 437** B

Root biomass included both live and dead roots. Crown biomass not included. Values followed by the same letter within a column are not significantly different at P=0.05. Values followed by the same capitol letter between columns within a depth increment are significantly different at P=0.05.

Since conversion from native shrub steppe to agricultural land five years ago, there has been a 20% increase (618 mg C kg^{-1}; 1232 kg C ha^{-1}) in soil organic carbon (SOC) in the 0-15 cm depth increment with no significant change in SOC below 15 cm. (For SOC data, as well as N, $\delta\ ^{13}C$ and percent of C derived from switchgrass (C_4) for 15 cm depth increments to 90 cm, see Appendix 22C). Over those five years, the land was planted to corn in 2002; potato in 2003, and has had monocultures of switchgrass from 2004 through 2006.

Total profile C (0-90 cm) among switchgrass monocultures increased 1758 kg C ha^{-1} above the uncultivated native soil (which was 18.5 Mg C ha^{-1}). Based on residue inputs calculated from production records and decomposition rates for potato and corn calculated in a separate study by Alva (2002), we estimated that 65% (1060 kg C ha^{-1}) of the average C increase (1620 kg C ha^{-1}) among cultivars in the surface 30 cm was derived from switchgrass (primarily from roots, since aboveground biomass was harvested). The other 35% of C increase was derived principally from the corn and potato residues. Our estimations are within the range of values found in the literature, though on the low end. Zan et al. (2001) found that switchgrass increased soil C by 3 Mg $ha^{-1}y^{-1}$ when compared to a corn field after 4 years of production. Lee et al. (2007) showed that the amount of C sequestered was dependent upon the type of N fertilizer applied. They reported a C sequestration potential of 2.4 Mg C $ha^{-1}y^{-1}$ with NH_4NO_3 fertilizers and 4.0 Mg C $ha^{-1}y^{-1}$ for manure-N within a 0.9 m profile of CRP lands in South Dakota. The addition of manure-C likely added significantly to the pool of stabilized C in this soil. Ma et al. (2000b) found little change in soil C after two years of production but found a 45% increase in 10 yr established switchgrass stands in Alabama.

The carbon "isotope fingerprint" data suggest that the switchgrass may be contributing more carbon than is evident from these measurements of differences in total soil organic carbon. For a technical description of the methodology and more detailed results, see Appendix 22C. The analysis showed a greater input of C (3368 kg C_4-C ha^{-1}) than that determined by the difference in mass of total C (1758 kg C ha^{-1}) between the uncultivated native soil and soils cropped to switchgrass. This suggests that over the three years of switchgrass cropping, ~1600 kg of carbon was replaced by new carbon in addition to the overall change in mass of soil C. The average accrual rate of C_4-SOC was estimated at 1.6 Mg CO_2e ac^{-1} y^{-1}, suggesting 30% of switchgrass roots turnover each year. This accrual rate was twice that of sequestration estimated by just the change in total C and is in line with the results of other research describing C sequestration by switchgrass (Liebig et al., 2008; Garten and Wullschleger, 2000; McLaughlin and Walsh, 1998).

Analysis of the CO_2 evolved during C mineralization was used to estimate the size and turnover rates for the active, slow, and resistant carbon pools. A discussion of these results, along with relevant data, is provided in Appendix 22C.

In interpreting these estimates of carbon sequestration under switchgrass production, it is important to remember that this study looked at irrigated switchgrass production on sites that prior to cultivation were naturally low in carbon. Switchgrass production for bioenergy will substantially increase SOC in soils that are by nature low in SOC; however, the outcomes may be different on sites that have higher initial C status.

Tradeoffs of Nutrient Removal under Switchgrass Production

Beyond the possible impacts on soil carbon, the harvest and removal of large amounts of biomass for biofuels production exports soil nutrients from the system (as does the harvest of any crop). We assessed the export of essential plant nutrients that occurred when the above ground switchgrass biomass was removed from the field, in order to determine impacts on soil fertility that will influence fertilizer recommendations. Fertilizer recommendations, in turn, will impact the overall GHG emissions balance for switchgrass production. And inadequate N inputs will ultimately prevent further soil C accumulation as soils maintain a relatively stable C:N ratio.

We looked at nutrient export in three different switchgrass cultivars (Kanlow, Shawnee, and Cave-In-Rock), and two different N application rates. The first N treatment was split application of 56 kg N ha-1, for a total annual rate of 112 kg N $ha^{-1}y^{-1}$, while the second treatment as split application of 112 kg N ha^{-1}, for a total annual rate of 224 kg N $ha^{-1}y^{-1}$. The first N application occurred in May prior to breaking winter dormancy and the second in July following the first of two annual harvests. Nitrogen and sulfur sources were urea (46-0-0) and ammonium sulfate (21-0-0-26); the P source was ammoniated phosphate (11-48-0).

To determine the nutrient export, we assessed the amount of dry biomass harvested from two annual harvests of switchgrass during 2005 and 2006 (one in late June/early July and one in October), after establishment of switchgrass in 2004. Nutrient concentration was sampled in above ground dry matter. We also harvested and measured root dry matter in 15 cm increments to a depth of 90 cm. A full description of methods is available in Fransen et al., (*in preparation*). The annual export of macronutrients from the field averaged 214 kg N ha^{-1}, 40 kg P ha^{-1}, 350 kg K ha^{-1}, 15 kg S ha^{-1}, 60 kg Ca ha^{-1}, 38 kg Mg ha^{-1}, and 6 kg Fe ha^{-1} among cultivars. Switchgrass required 1kg of N to produce 83 kg of biomass. Micronutrients (B, Mn, Cu, and Zn) removed at harvest averaged less than 1 kg ha^{-1} among cultivars.

Switchgrass may be a promising bioenergy crop for some areas of the state, specifically if it is used to restore degraded soils or in more novel cropping systems, such as companion planting in early successional hybrid poplar plantations. Analyses of the full environmental and GHG impact of switchgrass production for energy will need to consider whether the crop is really sustainable given the nutrient needs of the crop and the corresponding impacts on lifecycle GHG emissions. Economic analyses are also needed to determine whether or not the crop will provide a profitable option compared to other forage crops for growers in the PNW.

Tradeoffs in Policy Approaches to Biofuels

WSU and University of Idaho economists (McCullough et al., 2009) have also analyzed possible policy options for promoting biofuel usage in Washington State. Using a Computable General Equilibrium (CGE) approach, they examined the impacts of blend mandates, biofuel feedstock subsidies, fossil fuel taxes, and renewable fuel subsidies; tax approaches included both volumetric and carbon-based schemes in Washington State, a small, open economy. Impacts included state gross domestic product (GDP), net carbon emissions, and net economic welfare impacts for households (as measured by equivalent variation, EV).

Their modeling indicated that only blend mandates and taxes successfully reduced net carbon emissions. Taxes had a greater impact on lowering carbon emissions if they were carbon-based, rather than volumetric. While blend mandates were effective, they were also costly, and in general, modeling suggests they will only effectively catalyze growth of Washington agriculture if the feedstocks can be produced in this state at competitive prices.

Counter-intuitively, subsidies of biofuels are actually predicted to increase carbon emissions, because they lowered the overall price of fuel, and therefore, overall fuel consumption increased. They were also predicted to increase overall societal welfare as measured by equivalent variation, through their effect on lowering prices. In addition, feedstock subsidies were inefficient when growers had no comparative advantage (such as for corn), as this generally increased factor prices for competing crops that were more profitable, which decreased state GDP) from output declines, and negatively impacted household economic welfare as prices increased. In interpreting these results, it is important to note that the choice of policies will depend on the relative priorities of lawmakers: reducing carbon emissions, stimulating the Washington biofuels industry, or enhancing consumer well-being.

Meanwhile, researchers at the WSU School of Economic Sciences (WSU-SES) conducted an analysis of biofuels economics and policy that concluded that Washington is well positioned to be competitive in producing second-generation biomass-based fuels, many of which are discussed in the following section (Yoder et al., 2008). However, their analysis found that Washington State is unlikely to competitively produce large amounts of "first generation" feedstock crops, including corn, sugar, and oilseeds under most past and foreseeable market conditions (though this does not take account of potential crop advances made possible through ongoing research).

Based on these economic realities and an analysis of various possible policy responses, the WSU-SES suggests that if the State chooses to promote in-state production of biofuels, the most cost-effective approach would be to implement policy actions now that will prepare the state to take a leadership role in the advanced biofuels industry.

In the short run, if the State chooses to implement market incentives, they suggest that a carbon intensity tax may be the most effective way to lower GHG emissions and petroleum dependence and support the emergence of the biofuels industry. This policy would provide benefits to future as well as current technologies and thus would allow for flexibility in the development of the biofuels industry. This tax could be designed to be revenue neutral, and could generate a "renewable fuels fund" that would be available to fund tax credits for in-state production of fuels with low carbon intensities. Though other non-tax options such as renewable fuel standards or low carbon fuel standards may seem more appealing, their enactment would entail an implicit tax through their effects on fuel supply and demand, and would not as effectively encourage the development of biofuels with lower carbon impacts. Successful implementation of a carbon intensity tax would require the development of a region-specific database of life-cycle emissions for biofuels produced in the region.

Energy Production from Organic Wastes

Producing energy from waste materials is an attractive prospect, with substantial potential to mitigate greenhouse gas emissions, both through substitution for fossil energy products and through reduction of direct emissions that occur under current waste storage and treatment systems. Bioenergy produced from waste products tends to provide larger greenhouse gas benefits than bioenergy produced from dedicated crops, because these products avoid the large negative GHG impacts that are associated with the production of biofuel feedstocks, and also provide a net carbon gain in cases where the waste otherwise produces large emissions. However, this is not always the case, and it is important to evaluate the emissions impact on a case-by-case basis.

Energy can be produced from a range of organic waste materials. The CFF Project conducted considerable work with recovery of dairy waste that is covered in a separate section of this report (Chapters 2-12). We also carried out additional work in three areas that are particularly relevant to agricultural systems, with results that are presented in this chapter. First, in collaboration with the Washington State Department of Ecology, we carried out an inventory (Frear et al., 2005) and characterization (Liao et al., 2007) of biomass materials in the state, an essential first step towards developing a sustainable bioenergy policy, providing information useful for research prioritization and project feasibility analyses and prioritization. Second, we made some initial investigations of the impact of biochar on Washington agricultural soils (Granatstein et al., 2009).

Lastly, we evaluated potential environmental tradeoffs for the use of agricultural residues from dryland grain production systems for energy production. While our inventory included biomass estimates for crop residues, we believe that it is essential to more fully appreciate the important agronomic role of crop residues. Given this role, they should not be considered "wastes" available for energy production.

Biomass Inventory

A vast array of possible biomass materials can be converted into bioenergy with anaerobic digestion and other next generation biofuel technologies, including animal manures, forestry residues, food packing/processing waste, municipal wastes, and in some instances crop residues. While not all of these are agricultural in nature, a comprehensive assessment of the types and volumes of materials is a necessary first step to feasibility analyses and prioritizing investments in specific energy generation projects.

An inventory of Washington biomass (non-crop) carried out in 2005 in partnership with the Washington State Department of Ecology, geographically identified, categorized, and mapped 45 existing sources of organic materials in Washington at the county level that could potentially be used to generate energy. First, agriculture, processing and municipal statistics and databases along with personal interviews with agriculture and solid waste processing leaders led to development of an inventory of annual biomass waste streams. Second, the resulting biomass was standardized to represent total dry matter. Third, woody or straw-like materials with a high lignocellulosic content were evaluated for potential energy production using combustion as a conversion technology. Heat value coefficients were determined for each individual woody or straw-like material and used to calculate the potential electrical energy and power using a reference-based average of 20% conversion efficiency for non-combined heat/power combustion systems (CEC, 2004; Wilbur, 1985; Klass, 1993; Chartier, 1992). Fourth, the non-woody, wet biomass, represented largely by the animal manures and processing wastes, was evaluated for potential electrical energy production using anaerobic digestion as its representative conversion technology. In this process, dry biomass values were converted to available volatile solids and ultimately potential methane production using laboratory-determined coefficients for each of the biomass types. From the methane production levels, estimates of electrical energy and power production were developed using 30% conversion efficiency, also a reference-based average (CEC, 2004; Wilbur, 1985; Klass, 1993; Chartier, 1995). [3] Lastly, the biomass and bioenergy databases were mapped at the state and county levels on GIS and made web-accessible. For a more detailed description of methods, see Frear et al. (2005). To access the web-based GIS tool, visit http://pacificbiomass.org/WABiomassInventory.aspx.

The inventory indicates that 16.41 million dry tons of organic materials could be available for bioenergy production each year. This amount is significantly greater than the estimates of roughly 10 million tons for Washington State given in two national-level reports, the 1999 Biomass Feedstock Availability in the U.S. by the DOE-ORNL and the 2004 Billion Ton Report, indicating the importance of specific state inventories such as this one (ORNL, 1999; DOE, 2005).

Using rough calculations (with no consideration for material collection or process technology, etc.), this could theoretically generate more than 15.5 billion kWh of electrical energy or 1,769 MW of electrical power. Complete utilization of the inventoried biomass would therefore represent almost 50% of Washington State's annual residential electrical consumption (Haq, 2003). While it is unrealistic to think that all these materials would be converted to energy, such calculations indicate that a substantial portion of Washington's energy needs could be provided with significant potential reduction in GHG emissions through substitution for fossil fuels. A rough calculation (without consideration for life-cycle emissions) based on the carbon fraction of the inventoried biomass indicates that if 20% of the biomass was recovered and converted to energy annually, 10% of Washington's annual net GHG emissions could be mitigated.

Washington is blessed with a vast and diverse wealth of renewable biomass. The majority of the biomass (84.2%) is woody (lignocellulosic) material (Figure 22.5). Much of this biomass is forestry and crop residues that are quite dispersed and therefore are technically and economically challenging to collect and process. These

[3] Combustion of woody and straw-like materials, and anaerobic digestion of wet manures, municipal and processing waste were chosen as representative technologies because of their fit into the two main categories of waste and their simplicity for calculations. This should not be taken as an endorsement for either technology, or a rebuff of other technologies. Likewise, electrical energy production was the calculated product for this study, although numerous other products such as fuels and chemical bioproducts are possible. Any renewable energy initiatives will include multiple technologies and products, and any future studies and business plans will need to develop these options and evaluate "best fit" opportunities using appropriate social, economic, and environmental criteria.

residues also play important roles in maintaining soil health and therefore removal would raise significant sustainability concerns for soil health that are discussed later. However, some forms of woody biomass, such as mill residues and municipal yard and wood debris, are already concentrated. In addition, about 15 percent of the available biomass is in the form of more readily biodegradable and concentrated (but unfortunately also often lower energy conversion quality) waste streams coming from the Organic Fraction of Municipal Solid Waste (OFMSW), animal manures and food processing wastes.

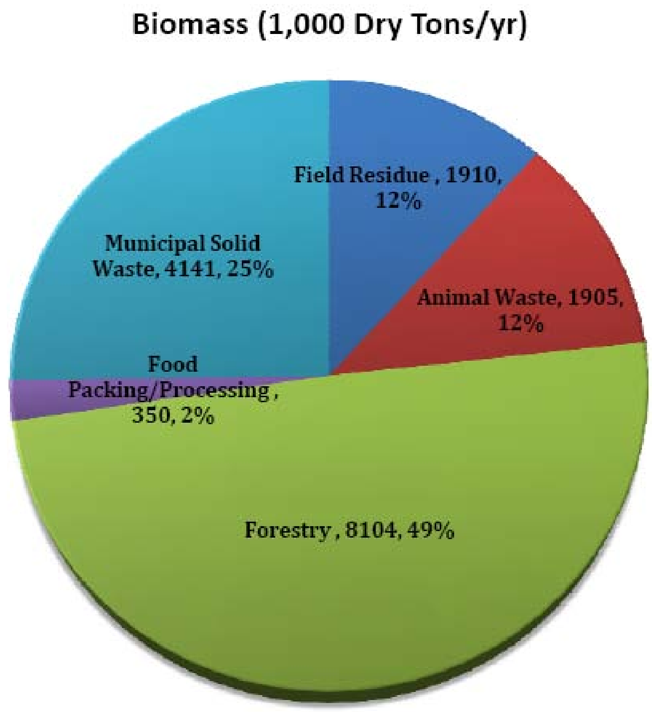

Figure 22.5. Biomass by category in Washington State.

The diversity of biomass materials opens the door to a potential bioproducts industry alongside a biofuels or bioenergy industry. Washington State's inventory shows much greater diversity in sources compared to inventories carried out in Midwest states. Based on this, Washington State could be well positioned to pursue a dual track which focuses on generating high value co-products from some of the concentrated, non-woody wastes while simultaneously devising collection and conversion capabilities for woody forestry materials.

Nutrient recovery is highly likely to emerge in conjunction with a bioenergy industry. Figure 22.6 indicates that amount of total nitrogen available by biomass category in the state. In 2001, Washington farms used a total of 176,000 dry tons of synthetic nitrogen fertilizer on an N basis, less than the total amount of nitrogen available in the inventoried biomass. While there are technical and economic obstacles that will prevent total recovery of this nitrogen, there are many opportunities to develop commercially viable biofertilizers, particularly from the most noxious “waste” biomass streams: manures and food processing wastes.

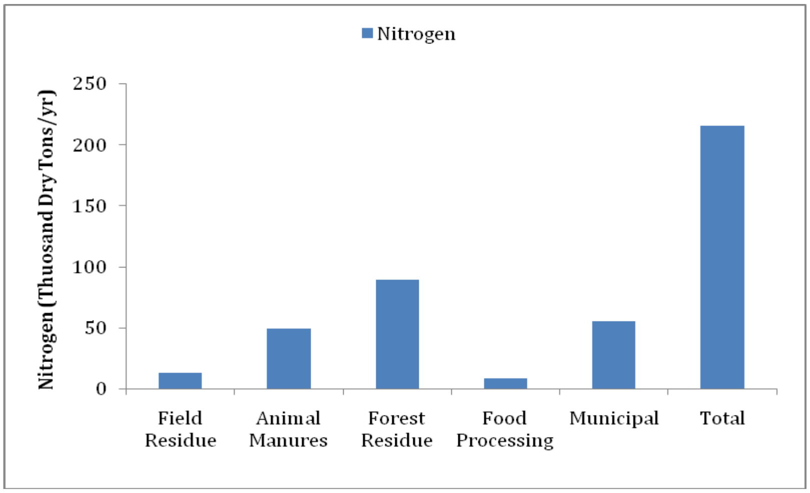

Figure 22.6. Nitrogen represented by the materials in the Washington State biomass inventory.

Because of extremely high transportation costs, biomass-related energy is most likely to be produced in or near areas with large, already collected biomass resources, specifically the materials that are “waste” products from animal feeding operations, food processing plants, and municipal waste facilities. Coincidentally, areas with these concentrated wastes align geographically with areas of the state where development of new business opportunities and jobs is of vital interest. Mapping of the biomass showed regional areas of concentration, focused in regions where forestry and municipal or forestry and agriculture intersect, such as the Puget Sound/Cascade and Yakima regions (Figures 22.7 and 22.8).

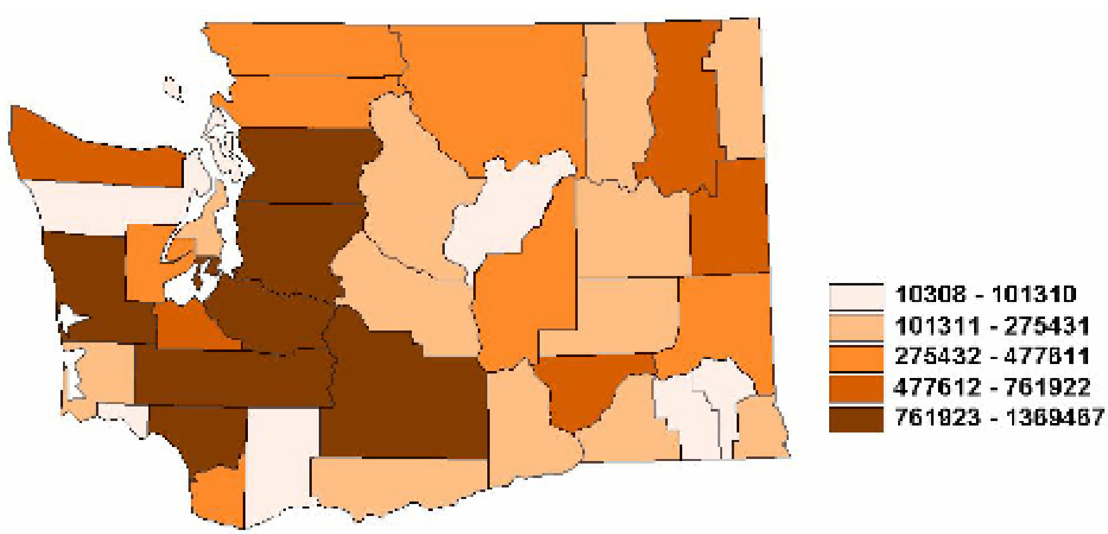

Figure 22.7. Biomass by County and Region (Biomass in dry tons)

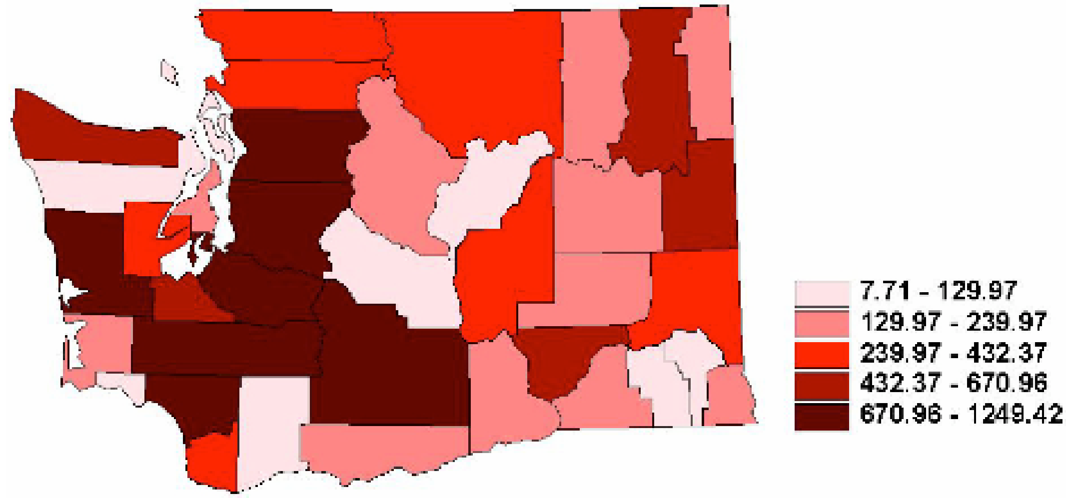

Figure 22.8. Bioenergy by County and Region (Bioenergy in M kWh)

Mill residues and municipal solid wastes (MSW) have a large influence on the result of this analysis. Therefore, because mill residue and MSW paper is already being successfully utilized for energy and recycling in many locations, we repeated the geographic analysis after eliminating these two categories of biomass (Figure 22.9). While this map is similar to the previous maps, it does make evident the agricultural and food processing strength of some of the counties that otherwise might not have been seen.

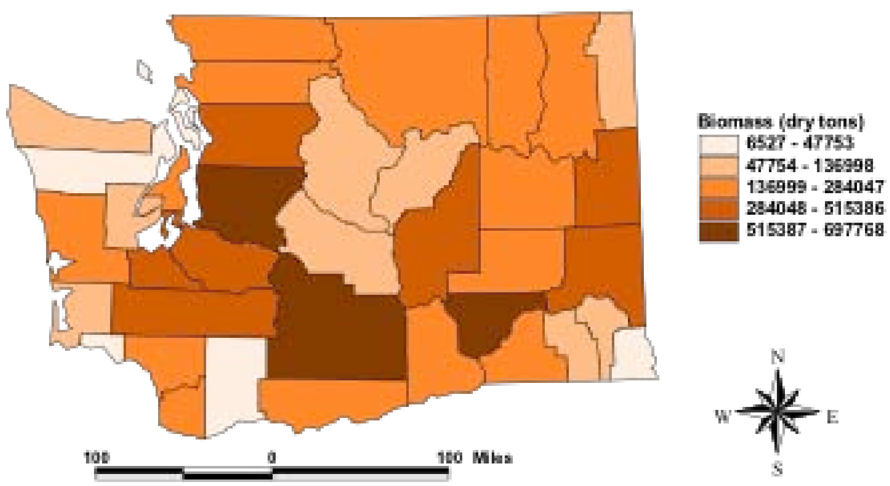

Figure 22.9. Biomass by County and Region without mill residue or MSW paper (Biomass in dry tons)

Distributed production not only is likely to be economically necessary; it also potentially creates other benefits such as decreased dependence on outside supply, increased market independence and local control. However, those using these data to develop policy or commercial investments that may project ten to twenty years forward should remember that the data presented are just a snapshot subject to future change. In addition, the inventory captured technical potentials, without regard to the policy environment or economic considerations. Technologies such as combustion, anaerobic digestion, and pyrolysis will likely be subject to local, regional, and state regulations.

Pyrolysis and Biochar

Pyrolysis and biochar have recently received increased interest for their potential to provide renewable liquid fuels and enhance carbon sequestration. Pyrolysis is a thermochemical conversion process best suited to dry, high carbon biomass. The biomass is heated in a low oxygen environment, where, rather than combusting, it forms three products: a gas, a charcoal-like solid called biochar, and a liquid sometimes called bio-oil. The gas is often used to provide the process energy, leaving the biochar and liquid as products. The liquid can be made into transportation fuels that substitute for petroleum-based fuels. The biochar, a very stable carbon-based material, can be added to soil for long-term C sequestration. If the biomass would otherwise be burned (e.g. grass seed straw), then emissions related to burning may also be avoided.

Biochar is resistant to microbial decomposition in the soil and can last for hundreds of years. It may also improve certain fertility aspects of soil, boosting crop yields. However, much of what is understood about biochar is derived from studies of *terra preta* soils in the Amazonian basin (Lehmann et al., 2004). Pre-Columbian civilizations once burned their refuse in these areas, leaving biochar-like materials that appear to have substantially altered soil physical and chemical properties and led to long-lasting carbon storage and improved crop production. There have been few studies of biochar in temperate climates, and the impact on agricultural soils in Washington is largely unknown. In addition, the production processes used by the Amazonians are not well defined, and it is not clear that adding biochar to soil leads to the equivalent of the *terra preta* soils. Biochar is a loosely defined term, and its performance in soil will likely depend on the feedstocks used, the processing temperature, the rates applied, and the crops grown.

In a preliminary assessment of the impacts of biochar on Washington agricultural soils, we tested biochars from five biomass feedstocks (pine chips, softwood bark, grass seed straw, and anaerobic digested manure fiber) produced at 500^{o}C with a pyrolysis unit developed by Washington State University, as well as peanut-hull biochar made at a pyrolyzer at the University of Georgia-Athens. Each biochar was compared to activated charcoal, and the impacts on five different soil types were evaluated for pH, buffering capacity, cation exchange capacity, water retention curves, soil nutrient availability (N,P, K, S, micronutrients), soil biological activity, and C sequestration potentials. Methods are described more completely in Collins et al. (2009).

Feedstocks differed in their chemical composition both before and after pyrolysis (for composition before pyrolysis, see Appendix 22D). Biochar from woody feedstocks had a higher C concentration than herbaceous feedstocks, generally above 75%, and a lower N content (Table 22.11). Most biochar from woody feedstocks had lower pH than biochar from herbaceous feedstocks and therefore would be of less liming value when applied to agricultural soils. For a full description of results, see Granatstein et al. (2009).

Table 22.11 Selected characteristics of the six biochars (after pyrolysis at 500°C) used in the laboratory analyses. Activated charcoal included as a standard analysis and comparison to biochars.

	Biochar Characters					
Source	C	N	S	C:N	C:S	pH
	-------------g/kg-------------					
Switchgrass	605 (26)†	20.6 (0.3)	1.8 (0.4)	29	336	9.4
Digested Fiber	667 (16)	22.3 (0.4)	2.9 (0.3)	30	230	9.3
Peanut Hull	706 (12)	17.4 (0.9)	0.6 (0.1)	41	1178	9.6
Bark	745 (4)	3.4 (0.3)	0.3 (0.1)	219	2483	7.6
Soft Bark	778 (7)	4.4 (0.3)	0.6 (0.2)	177	1482	8.4
Pine Pellets	800 (8)	1.4 (0.3)	0.4 (0.1)	571	2000	7.4
Activated Charcoal	873 (3)	4.7 (0.6)	7.6 (0.4)	186	115	9.1

†Standard error of mean in parentheses.

The concentration of recalcitrant carbon in biochar was determined using acid hydrolysis, which removes more labile forms of C, leaving an acid-resistant fraction that has been shown to have a mean residence time of 100's to 1000's of years (Collins et al., 2000). This is the fraction that would likely be eligible for carbon credits. Herbaceous materials (switchgrass and digested fiber) lost 6-8% of their total C and <0.2% of their N after acid hydrolysis, while the woody feedstocks (softwood bark and wood pellets) remained largely unchanged (Table 22.12). We believe that the C loss originates from condensates of the bio-oil coating the biochar following pyrolysis. It is unclear why we did not observe the loss with the woody feedstocks.

Table 22.12. Selected characteristics of the six biochars (500°C) after acid hydrolysis. Activated charcoal included as a standard analysis and comparison to biochars.

	Biochar Characters After Acid Hydrolysis*					
Source	C	N	S	C:N	C:S	pH
	-------------g/kg-------------					
Switchgrass	656(9)†	18.1 (0.3)	1.8 (0.5)	36	364	nd
Digested Fiber	731 (27)	23.8 (0.2)	4.1 (0.4)	31	178	nd
Peanut Hull	Nd	nd	nd	nd	nd	nd
Bark	758 (2)	3.7 (0.1)	1.8 (0.3)	205	421	nd
Soft Bark	Nd	nd	nd	nd	nd	nd
Pine Pellets	800 (3)	1.5 (0.3)	1.6 (0.1)	533	500	nd
Activated Charcoal	873 (3)	4.7 (0.3)	8.0 (0.3)	186	109	nd

†Standard error of mean in parentheses. Nd –not determined. *Acid hydrolysis is a method used to determine the concentration of recalcitrant C. 6N HCl contains 1.2% sufur.

The biochars were added to five different soils from Washington State, representing the diversity of agroclimatic regions and important crops:

Quincy sand. Young alluvial soil, low nutrient and water holding capacity, found in central Washington. Crops grown under irrigation. Common crops: potatoes, corn, wheat, alfalfa, apples.

Naff silt loam, Palouse silt loam, and Thatuna silt loam. Three different types of soil formed from loess deposits under grassland in eastern Washington. Dryland farming, annual cropping. Common crops: wheat, barley, peas, lentils.

Hale silt loam. Soil formed from loess and volcanic ash over glacial outwash in western Washington. Rainfed cropping, some summer irrigation, seasonal waterlogging. Common crops: hay, corn silage, and other forages for dairy cows.

These soils represent a range in terms of soil organic matter (C), pH, and cation exchange capacity (for data, see Appendix 22D). Each of these characteristics could potentially be influenced by biochar amendment.

Soils were amended with three rates of biochar (0.4% by mass, or 9.8 metric tons (Mg) per hectare; 0.75% by mass, 19.5 metric tons/ha; 1.5% by mass, or 39.0 metric tons/ha). Biochar addition did impact soil characteristics, and in some cases had different impacts in different soil types. Soil pH was found to increase 1 unit for the highest rate of biochar addition for the herbaceous feedstocks, and 0.5-1.0 units for the woody sources. The increase in soil C and N after biochar additions followed the pattern of Hale<Thatuna<Palouse<Naff<Quincy, the result of high background soil C for the silt loams versus low soil C for the sand. While cation exchange capacity (CEC) was not significantly impacted, soil water holding capacity was increased in the Quincy sand. For detailed data on the influence of biochar on soil pH, CEC and water holding capacity, as well as C, N, and S recovered after amendment, see Appendix 22D.

In general, all biochars on all soil types increased soil C with increasing application rates, as shown by the nearly linear relationship between the amount of a C added as biochar and the amount of additional C recovered measured in the soil after amendment (Figure 22.10). We found a similar response for soils containing biochar following acid hydrolysis (Figure 22.11). This stable C pool comprised between 60 - 90% of the total soil C depending on soil type (see Appendix 22D for data). The size of this pool indicates the recalcitrance and persistence of biochar in soil. C-mineralization studies confirmed that the majority of the C added was biologically inert, and that the addition of biochar did not accelerate loss of indigenous organic matter through the 'priming effect' (see Appendix 22D for data).

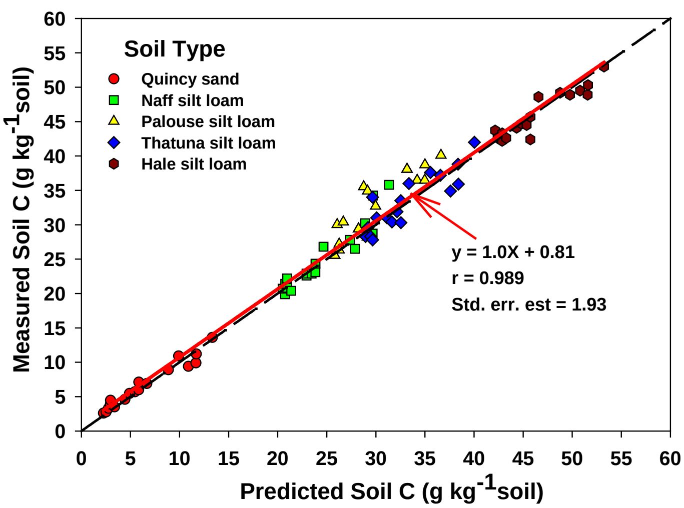

Figure 22.10. Comparison between the amount of C added in the biochar amendments and the amount of additional C measured in the soil after amendment.

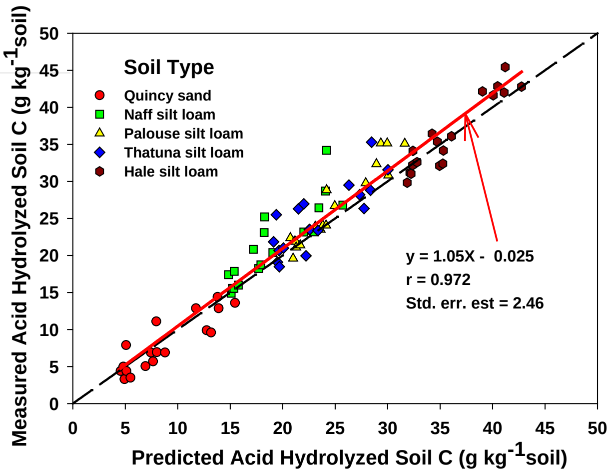

Figure 22.11. Comparison between the amount of C added in the biochar amendments and the amount of additional C measured in the soil after acid hydrolysis.

N-mineralization studies were carried out to explore the impacts of biochar on forms of nitrogen that are available to plants. The results showed a consistent decrease in nitrate production with increasing rates of biochar, across all types of biochar and soils (for data, see Appendix 22D). This suggests that the N contained in the biochar is locked into the carbon matrix and not available to microorganisms or plants.

A greenhouse study tested the impact of softwood bark and wood pellet biochars on wheat grown in each of the five soils (vegetative growth only). There were no significant differences in total wheat biomass due to biochar rate for any of the biochars amount soil types (Table 22.13). However, though it was not statistically significant, total plant biomass tended to increase with increasing biochar application rate, except for the amendment rate of 39 Mg/ha, which tended to lead to a reduction in growth.

Table 22.13. Wheat roots, shoots and root:shoot ratio after growth in soils amended with the softwood bark and wood pellets biochars.

Soil Series	Plant Characters					
	Biochar	†Rate	Root	Shoot	Total	R:S
		Mg ha^{-1}	--------- g ---------			
Quincy	Softwood bark	0	12.3 (3.6) a	10.3 (3.1) a	22.6 (5.0) a	1.19
		9.8	11.3 (3.6) a	10.3 (3.1) a	21.6 (5.9) a	1.10
		19.5	10.5 (3.8) a	12.2 (2.9) a	22.7 (5.9) a	0.86
		39.0	8.6 (3.5) a	9.1 (1.6) a	17.7 (5.0) a	0.95
	Wood pellets	0	18.5 (7.3) a	8.3 (1.8) a	26.8 (9.6) a	2.22
		9.8	14.0 (9.7) a	9.7 (2.5) ab	23.7 (11.7) a	1.44
		19.5	12.1 (5.5) a	12.4 (1.9) b	24.5 (7.1) a	1.00
		39.0	11.2 (3.1) a	8.2 (2.2) a	19.4 (4.5) a	1.37
Naff	Softwood bark	0	7.3 (1.5) a	5.1 (1.0) a	12.4 (1.8) a	1.43
		9.8	12.9 (3.8) b	4.9 (1.3) a	17.8 (4.9) a	2.63
		19.5	13.9 (4.4) b	5.9 (1.0) a	19.8 (4.8) a	2.36
		39.0	12.6 (5.7) ab	3.8 (1.5) a	16.4 (6.2) a	3.32
	Wood pellets	0	7.3 (1.5) a	5.1 (1.0) a	12.4 (1.8) a	1.43
		9.8	5.0 (1.6) ab	4.7 (1.1) a	9.8 (2.3) a	1.06
		19.5	6.1 (1.0) ab	7.0 (0.7) b	13.1 (1.6) a	0.87
		39.0	4.6 (1.0) b	4.0 (1.6) a	8.6 (2.4) a	1.15
Palouse	Softwood bark	0	8.7 (2.1) a	6.8 (1.3) a	15.5 (3.0) a	1.28
		9.8	8.4 (2.9) a	7.0 (1.2) a	15.3 (3.2) a	1.20
		19.5	11.8 (3.4) a	12.3 (1.7) b	24.1 (4.5) b	0.96
		39.0	8.6 (2.2) a	8.7 (2.2) ab	17.4 (4.2) ab	0.99
	Wood pellets	0	11.7 (2.5) a	4.8 (1.0) a	16.6 (3.2) a	2.44
		9.8	10.0 (2.3) a	4.3 (1.0) a	14.3 (2.8) a	2.33
		19.5	13.2 (3.0) a	7.0 (1.9) a	20.2 (4.8) a	1.88
		39.0	9.3 (2.0) a	4.0 (1.2) a	13.4 (2.8) a	2.33
Thatuna	Softwood bark	0	14.8 (2.4) a	4.8 (1.4) a	19.6 (4.6) a	3.10
		9.8	19.4 (6.4) a	5.7 (1.3) a	24.1 (7.8) a	3.40
		19.5	24.5 (8.6) a	7.9 (2.9) a	32.4 (11.3) a	3.10
		39.0	15.6 (4.9) a	4.7 (1.2) a	20.3 (5.3) a	3.32
	Wood pellets	0	10.9 (2.4) ab	4.8 (1.4) a	15.7 (3.5) ab	2.27
		9.8	10.2 (2.9) ab	5.1 (1.2) a	15.3 (3.8) ab	2.00
		19.5	14.6 (4.4) a	7.6 (1.1) b	22.1 (5.1) a	1.92
		39.0	6.7 (1.8) b	4.6 (1.0) a	11.3 (2.5) b	1.46
Hale	Softwood bark	0	10.1 (3.9) a	7.3 (2.7) a	17.4 (5.4) a	1.38
		9.8	11.4 (2.6) a	9.1 (1.5) a	20.5 (3.5) a	1.25
		19.5	12.9 (3.0) a	11.6 (2.6) a	24.5 (5.2) a	1.11
		39.0	10.3 (2.8) a	9.1 (2.6) a	19.4 (5.1) a	1.13
	Wood pellets	0	10.1 (3.9) a	7.3 (2.7) a	17.4 (5.4) a	1.38
		9.8	9.3 (1.6) a	6.3 (2.1) a	15.6 (3.0) a	1.48
		19.5	11.3 (4.7) a	7.5 (1.5) a	18.9 (5.8) a	1.51
		39.0	9.7 (2.2) a	5.2 (1.4) a	14.9 (2.2) a	1.86

‡Std. error of mean in parentheses. Statistical comparisons were not made among biochars because the wheat was not grown at the same time. Values for a biochar within a column followed by the same letter are not significantly different at $p = 0.05$.

These studies present initial findings about the potential role of biochar in our region's agricultural soils. How biochar's impacts on soil may change over time as it integrates with the soil matrix is unknown. Expanding research on possible crop responses (including field trials), and, where crop responses are seen, investigating the mechanisms responsible, will begin to develop a body of knowledge that will allow for predictions of biochar's efficacy.

In addition, other possible effects, both positive and negative, should be considered. Some have suggested that biochar may have more positive impacts in combination with other organic amendments such as compost or brassica seed meals, while others have suggested various novel uses of biochar to recover nutrients from water (e.g. livestock lagoons, drainage canals), or for seed-zone pH adjustment. Meanwhile, preliminary studies by R. Boydston (USDA-ARS, Prosser, WA) indicated that biochar did interact with two herbicides tested, and this would need to be considered by farm managers.

Tradeoffs of Crop Residue Removal

Biomass materials such as crop residues, OFMSW, and forest residues can be used for cellulosic biofuels, and the assumption that all of these feedstocks are "wastes" is a fundamental assumption leading to the prediction that cellulosic biofuels will have a more beneficial life-cycle GHG and net energy impact than current ethanol biofuels. However, not all of these materials can be considered "wastes" in the sense that there may be significant ecological benefits forgone if these materials are collected and converted into biofuel. Many of biomass inventories (e.g. ORNL's Billion Ton Report - http://feedstockreview.ornl.gov/pdf/billion_ton_vision.pdf) don't give sufficient consideration to the organic materials they characterize as "wastes", specifically crop residues.

In order to contribute to better public understanding of this set of issues, we assessed the consequences associated with different crop residue management options using field-scale research conducted at the Washington State University Cook Agronomy Farm near Pullman, Washington. In particular, we evaluated the impacts of wheat straw residue removal for energy production on soil carbon sequestration and crop nutrient removal.

In the dryland cropping region of the Inland Pacific Northwest, winter wheat yields often exceed 6725 kg/ha (100 bushels/ac) and associated crop residues following harvest can be in excess of 11,209 kg/ha (10,000 lbs/ac), equivalent to as much as 3742 L/ha (400 gal/ac) of ethanol with current technology if all residues are removed. On average, though, residue production (and the associated ethanol yield) is generally much lower (Figure 22.12).

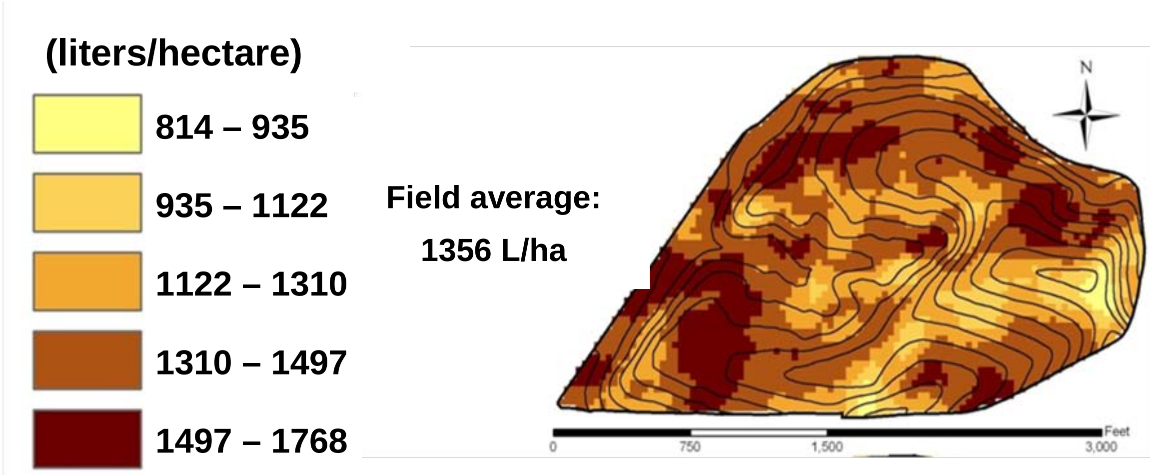

Figure 22.12. Potential ethanol production from wheat straw remaining after harvest of 6053 kg/ha winter wheat, Cook Agronomy Farm, Pullman, WA.

Using these optimistic residue yields of 11,209 kg/ha, many have suggested that residue can be removed without detrimental impacts on soil carbon levels. Crop residues are composed of about 40 to 45% carbon; therefore, 11,209 kg/ha of residues contains about 4,483 to 5044 kg/ha of C. Considering that carbon inputs of about 2,250 kg/ha are needed to sustain current levels of soil carbon in typical agricultural lands, it appears at first glance that removal of some of the residue for bio-energy is an attractive option with little downside. As is often the case, however, the actual situation is more complicated. When the information from this calculation is applied generally, several questionable assumptions have to be made, notably that:

(1) current soil carbon levels are optimal or at least adequate and sufficient for sustained agricultural production and fulfillment of ecosystem services;

(2) production levels of crop residues are similar for a given field every year; and

(3) production levels of crop residues are uniform within the same field in any given year.

The first assumption is inaccurate for most dryland agricultural soils around the world, including the Inland Pacific Northwest. Reliance on mechanical cultivation (e.g. moldboard plowing, disking, etc.) coupled with reduced organic carbon inputs has historically led to exponential declines in soil organic matter following the conversion of native prairies to agricultural production. In the Inland Pacific Northwest, current soil carbon levels are about 50% of the original levels found under native prairies, indicating that historical soil management practices have contributed to atmospheric carbon increases. Severe soil erosion, which preferentially removes soil organic materials from farm fields, has also contributed to declining soil carbon with eroded soils often averaging only 25% of native soil carbon levels. Figure 22.13 shows the differences between native and cultivated conditions for three characteristic soil types in the Palouse.

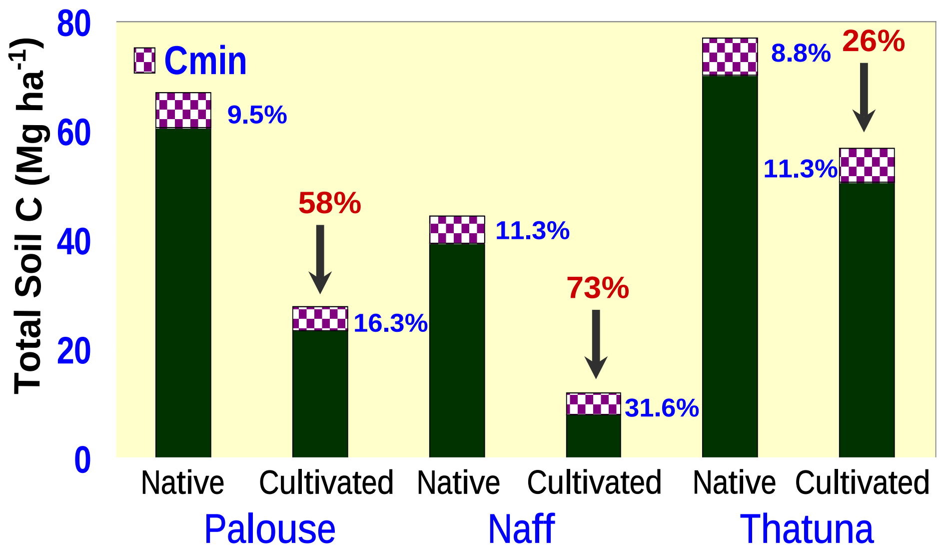

Figure 22.13. Total soil carbon and portion mineralized (C min = C mineralized in 182 day incubations) for Palouse, Naff and Thatuna soils (0-20 cm) at neighboring native and cultivated sites.

The loss of soil organic matter has seriously degraded the inherent productivity of many soils by reducing water infiltration, water holding capacity, nutrient supplying power and effective rooting depth. Degraded soils require increased inputs of nutrients and water to maintain crop yields and often are more vulnerable to further degradation under normal weather extremes. Advances in cultural practices, such as no-till seeding technology, have enabled farmers to slow carbon losses and in some cases to actually regain some soil carbon. Reversal of long-term negative trends in soil organic carbon, however, has only been possible because all crop residues have been returned to the soil.

While reaching native levels of soil carbon is an unrealistic goal for most farms, continued increases in soil carbon of degraded soils will provide improvements in agricultural productivity, decreased reliance on synthetic fertilizers that are energy and GHG-intensive to produce, and increased carbon sequestration (one kg of stored soil carbon is equivalent to 3.67 kg of carbon dioxide).

The assumption that production levels of crop residues are similar for a given field every year is also inaccurate. It is difficult to raise high-yielding crops like winter wheat on the same field each year in the dryland regions of the PNW due to limited water availability and to increasing pressures from various weeds, diseases and insects. Therefore, to enhance the production of wheat, rotations and fallow periods are used to break up disease and weed cycles and enable the soil to store water.

Common crop rotations consist of winter wheat grown once every two or three years with the remaining years consisting of fallow or rotational crops such as spring barley, grain legumes and canola. These rotational crops produce much less crop residue than winter wheat, quantities that are insufficient to maintain soil carbon levels. Fallow periods also reduce the total amount of residue available over the crop rotation. Consequently, to maintain soil carbon levels in a given field, the relatively high amounts of residue produced during the winter wheat rotation must compensate for years in which residue production is insufficient. Therefore, the amount of winter wheat residue that is actually available for removal is much lower than it seems at first glance.

The last assumption, that production levels of crop residues are uniform within the same field in any given year, creates a special concern for decision-making in the Pacific Northwest due to the highly variable in-field topography and soil types in our region. It is common for a single field to have 3-fold to 4-fold variability in yield in a given year due to differences in soil carbon levels, slope and aspect. As a result of this variability, uniform management strategies applied to a single field will have dramatically different impacts in different parts of the field. Therefore, it may be reasonable to remove substantial residues from one part of a field without creating negative impacts, while removing any residue from another part of the field may be extremely detrimental.

Figure 22.14 below provides an illustration showing the expected availability of wheat residue carbon from a three-year crop rotation (winter wheat –spring pea – spring wheat) at the WSU Cook Farm in Pullman. From this image, the in-field variability of residue production is obvious. Note also that average residue production for the field during the spring pea year is only 981 kg C/ha, much lower than amount needed to maintain soil carbon levels over the rotation (2250-2750 kg C/ha). The field average production of winter wheat residues (3431 kg C/ha) makes up for this on average, but only if no residues are harvested. And even without any residue harvest, there are areas of the field which likely are not receiving enough residues to maintain carbon levels over time.

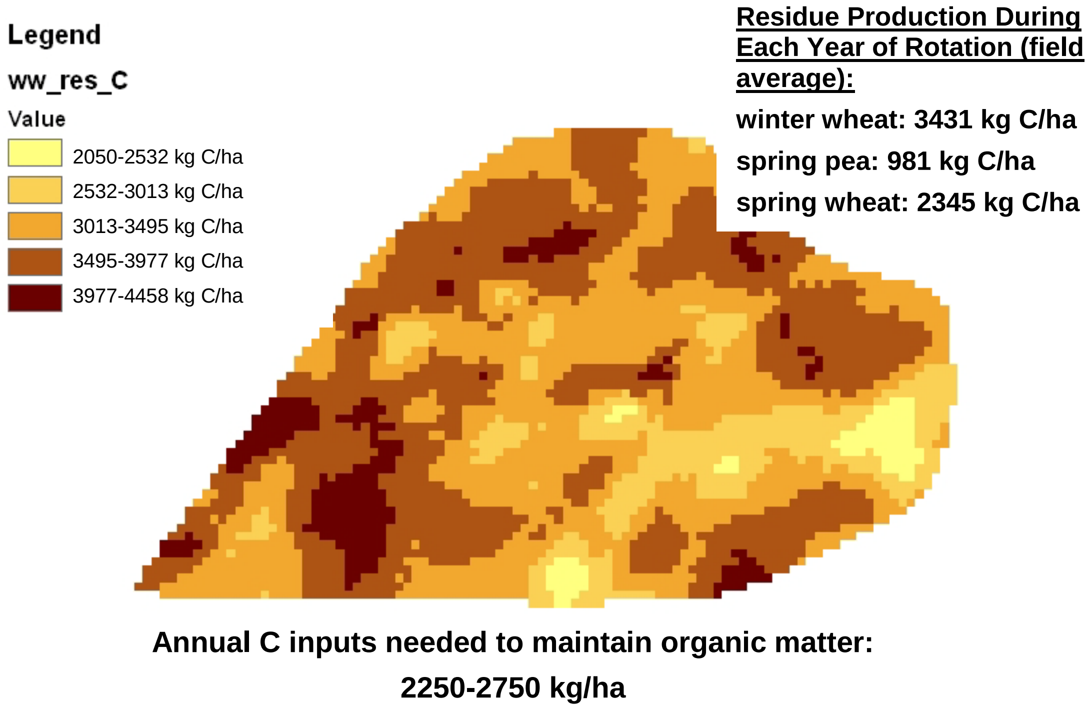

Figure 22.14. Carbon (kg/ha) remaining in winter wheat residues produced in a winter wheat, spring pea, spring wheat rotation at WSU Cook Agronomy Farm, Pullman, WA.

In addition to impacts on soil carbon and the associated consequences for erosion and microbial activity, removal of crop residue has substantial impacts on soil nutrient cycling and availability. Figure 22.15 shows the amount of N, P, K and S removed from the field if wheat straw were removed for energy production. While these nutrients could be replaced with synthetic fertilizer if prices were high enough to compensate, according to June 2007 input costs, the value of nutrients removed was greater than $13 per ton of wheat straw.

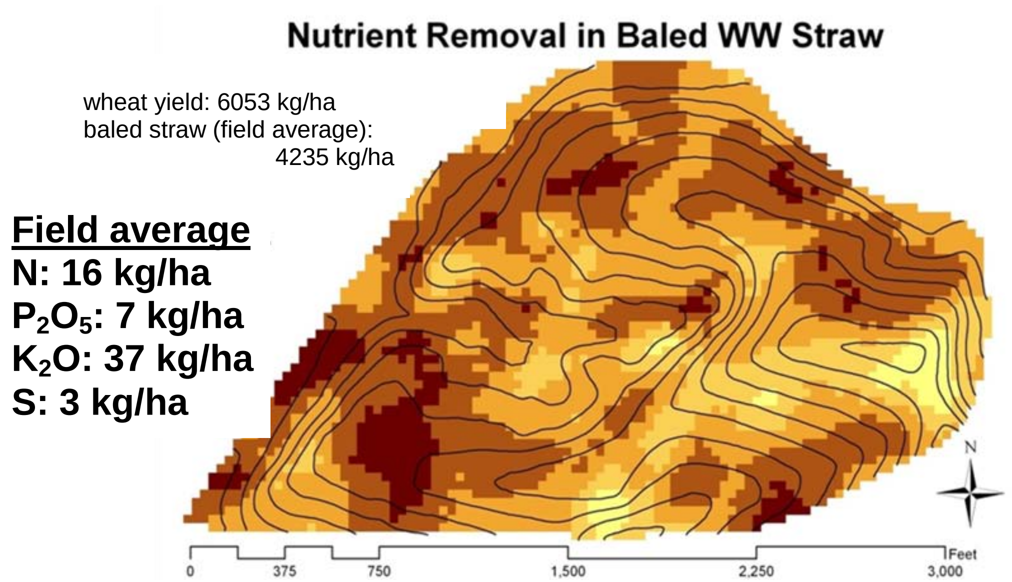

Figure 22.15. Nutrient removal from residue removal, WSU Cook Agronomy Farm, Pullman, WA.

As this analysis shows, incorporating a more comprehensive understanding of the current agronomic, ecological, and economic value of crop residues into biomass inventories may show that the real levels of residue available for energy production on a sustainable long-term basis may be much lower than anticipated. Analyses such as these are critical for developing realistic cellulosic ethanol projects and sustainable bioenergy plans.

Conclusion

The agricultural sector has the potential to contribute to greenhouse gas mitigation by producing energy products that reduce emissions in the energy sector by displacing fossil fuel products. However, economic analysis shows that Washington State is unlikely to become a major producer of first generation biofuels such as oilseeds. Despite this, these crops may have an important role to play as an alternative to other rotational crops, if they can be incorporated into existing rotations. On the other hand, Washington is well positioned to become a leader in second-generation biofuels. Promising feedstock possibilities include switchgrass (or possibly other perennial grasses), animal manures, food processing wastes, and OFMSW. Despite the promise, each of these potential biofuels has potential drawbacks, and more complete assessment of net GHG and other environmental impacts will be essential to developing effective GHG mitigation policy that includes regionally produced biofuels.

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Appendix 22A. Crop Descriptions and Cultural Considerations for Oilseeds in the Pacific Northwest.

H. Collins, A. Hang, and R. Boydston

Canola/Rapeseed

Canola has a potential role to play in many of the dominant cropping systems in the state, including dryland eastern Washington, irrigated areas in the Columbia Basin, and western Washington. In dryland eastern Washington and nearby areas of Oregon and Idaho, where wheat is the most profitable crop, canola can provide rotational benefits to dryland grain growers. Canola breaks up disease and insect cycles. It also can help with weed control, as growers can utilize different herbicides than on wheat, particularly those that target unwanted grasses. However, the crop does have some disadvantages. Canola is more difficult to establish than some other rotational crops, and may require more moisture. In addition, high summer temperatures can cause premature flowering and seed abortion in spring-planted cultivars, resulting in low yields.

Canola and rapeseed are small oilseeds in the *Brassica* family. Canola is similar to rapeseed but has an edible high quality oil and ready-to-feed high quality meal because of its low erucic acid and glucosinolate content compared to rapeseed. Both canola and rapeseed have cultivars that can be spring or fall planted. Spring canola is planted in late March or early April and is ready for harvest 100 to 120 days later, with yields ranging from 1500 to 2000 lbs ac^{-1} under irrigation. Winter canola or rapeseed is planted in late August or early September depending on location, and is harvested about 10 months later (June or July). Yields are normally double the yield of the spring sown cultivars. Equipment used for planting and harvesting is similar to that used in small grain production.

When seedbed conditions are optimal, a seeding rate of 4 to 6 lb per acre is recommended for canola. Seed needs to be in firm contact with the soil at a uniform depth of ½ to 1 inch deep. In less than ideal planting conditions, a seeding rate of 6 to 8 lbs is recommended. Treated seed costs considerably more than untreated seed, but studies have shown that the treatments are cost-effective (Painter, 2006). An ideal row spacing is from 6 to 7 inches apart.

Both spring and fall cultivars use 10 to 12 inches of water. Canola and rapeseed have deep root systems and can use soil moisture left below the rooting zone of previous crops (e.g. potatoes). Winter canola normally uses less irrigation water than spring canola, as it uses residual soil moisture accumulated during the winter months.

Canola provides several benefits to the agricultural ecosystem. Winter canola has been shown to protect soil from erosion during the winter, especially in hilly areas with high precipitation. The deep root systems of winter canola/rapeseed break down hard pans and improve uptake of nutrients leached below the root zone of previous crops, reducing ground water contamination. In addition to these benefits, growers will return more organic residues to the soil enhancing soil organic matter.

While low temperatures are generally not a problem in our area (yield losses from late frosts have not been reported), canola is sensitive to heat and drought stress during flowering. For spring cultivars in particular, high summer temperatures can cause bolting, seed abortion and lowered yields. Early establishment of winter canola or rapeseed improves bloom set allowing the crop to escape the high summer temperatures that can cause seed abortion.

Meeting fertility requirements is important for ensuring optimal yields, and testing of the entire profile is recommended. Canola requires about 6.5 lb of nitrogen for each 100 lb of seed yield. A 6:1 ratio of nitrogen to sulfur is recommended, with at least 10 lb per acre of available sulfur (more detail on optimal nitrogen application by soil moisture and expected yields can be found on the Canadian Canola Council's extensive website, http://www.canola-council.org/nitrogenintro.aspx). Recommendations for phosphorus depend on field history, but there should be about 10 lb available phosphate per acre.

Canola is sensitive to burning from nitrogen placed in the seed row. An effective strategy for avoiding this problem is deep-banding fertilizer in the fall. For dry soils, this also increases the availability of nutrients in the root zone, rather than trapping them in the upper dry layer of soil.

Canola and rapeseed are not hosts for, or susceptible to many of the pests and diseases that are common in wheat crops, including Russian wheat aphid, Hessian fly, take-all (*Gaunmanomyces graminis*) and eyespot (*Pseudocerospoelle hepitricoides*). Thus, when canola or rapeseed is grown in rotation with wheat, it reduces the levels of these pathogens in subsequent wheat crops.

Pests of canola and rapeseed include flea beetles, aphids, and the cabbage seedpod weevil. However, because canola matures relatively early (in late spring/early summer), infestations of aphids are present but are generally not a serious problem. The most serious pathogen is Sclerotina white rot.

Harvesting can be done directly with a combine. However, some producers find that this causes problems with seed shattering. To counteract this problem, some producers swath canola before combining (though this adds expense and is time-consuming). Polymer sprays are also available to counteract shatter problems, and are more commonly used on winter canola varieties.

In the past, Washington-grown rapeseed/canola was shipped out of the area for processing. Recently, a crushing facility was under construction in Eastern Washington. When completed, the plant is expected to produce oil for the biofuels industry and other uses, increasing local demand for oilseeds.

Local oilseed crushing facilities will also open new markets to growers, as they will produce a variety of by-products. Oil-free canola meal is in high demand for livestock feed in our area, which imports the majority of animal feed protein requirements, High glucosinolate meal, another possible byproduct, is a biopesticide with the potential to benefit organic farming and/or the horticultural industry.

Additional value-added products that may be developed include formulated feeds, proteins, and amino acids, which can be marketed locally.

Mustard

Mustard (*Brassica spp.*) can tolerate drought better than canola or rapeseed, and can be produced on marginal soils, though it does respond to fertilizer and water applications (Figure 22.A1). However, it yields less oil than canola or rapeseed (25 to 30% compared to 40 to 45%), and unlike canola, the meal normally contains high concentrations of glucosinolate compounds, making it inappropriate for animal feed.

Figure 22.A1. Mustard.

While these compounds lower the value of the meal byproduct, they also provide pest-inhibiting properties that can benefit rotational cropping systems. Over the past four years the Integrated Farming Systems group at Prosser, WA has been recommending that growers incorporate oilseed cover crops that contain glucosinolates in rotation to control soil pathogens and protect soil resources. Research by Andy McGuire (WSU Extension) showed that incorporating a mustard cover crop can offset soil fumigation costs by up to $100/ac. As a result, the area planted to mustard oilseed cover crops in the Columbia Basin has increased from 400 to 20,000 acres, and is continuing to increase.

Currently, mustard cover crops and other oilseed green manures are planted and incorporated in the fall prior to reaching seed maturity. However, over the last three years we have maintained a series of trials evaluating oilseed crops grown to maturity to determine how they can fit into current high value irrigated vegetable cropping systems. We hope that this research will help improve farm profitability while maintaining the desired benefits of biofumigation. Like canola and rapeseed, spring planted mustard varieties reach maturity in 100 to 120 days, while fall planted cultivars are ready in about 10 months

We are also continuing to investigate other possible uses for the meal, in order to improve overall crop profitability. High glucosinolate mustard meal can be used as soil fumigant, suppressing nematode and weed populations in organic production systems or the horticultural industry.

Safflower

Safflower (*Carthamus tinctorius)* belongs to the Compositae family and can be used for food, flower arrangements, medicine or dyes (Figure 22.A2). Safflower can tolerate extreme weather conditions, and is considered a low input and drought tolerant crop, though it responds well to irrigation and fertilizer. It is planted in early spring and reaches maturity in about 5 months in Washington. Seed yield is about 3000 to 3500 lbs with oil concentration of 42 to 48% depending on the variety.

Figure 22.A2. Safflower.

Soybean

Soybean (*Glycine max*) grows very well in the PNW with irrigation if the proper maturity group is chosen (Figure 22.A3). In the South Columbia Basin, early varieties of maturity groups 00, 0 or 1 produced well. A Rhizobia inoculant is required for producing high yields, particularly where soybeans have not been grown before. Soybean yields range from 3500 to 4000 lbs/acre under irrigation. Soybeans have a lower oil concentration (15-20%) than canola or rapeseed. However, the meal byproduct is very high in protein and thus is a high quality feed for farm animals.

Figure 22.A3. Soybeans.

Other Oilseed Crops

Several other oilseed crops such as meadow foam, camelina and crambe have been grown in central Washington but weed problems and low economic returns make them a high risk at this time.

Appendix 22B. Crop Description and Cultural Considerations for Switchgrass in the Pacific Northwest.

H. Collins, S. Fransen, and R. Boydston

Switchgrass (*Panicum virgatum*) is a warm-season, deep-rooted perennial grass species with the potential to supply cellulosic ethanol production (Figure 22.B1). While not native to the Pacific Northwest, it is a viable crop in the warmer regions of the PNW if natural rainfall is adequate or if irrigation water is applied. While switchgrass has not until now been grown in the region as a biofuel crop, the grass has been successfully grown as a seed crop for more than 20 years.

Figure 22.B1. Switchgrass

In the irrigated regions of the PNW, switchgrass should be planted by late May to mid-June. Switchgrass seed is small with about 325,000 seeds per pound, and naked, making it easy to drill. Seeds should be planted into a clean and firm seedbed with a drill using a covering chains or packing wheels to ensure good soil-seed contact for rapid germination. We have successfully established stands with seeding rates from 7 to 12 pounds pure live seed per acre with a drill on 6-inch centers. It is important to check the seed tag for high seed germination rate, as this percentage can vary widely among varieties and seed sources.

Switchgrass is slow to germinate and all switchgrass varieties start as a bunchgrass, though they develop into sod over time with proper management. In combination, these two characteristics mean that controlling weeds is particularly critical during the first year of establishment. A combination of chemical and cultural controls are usually necessary, as new weeds can germinate all season long from routine irrigation, particularly if the weed seed bank is high, Prowl (pendimethalin) applied pre- or post-emergence at 0.66 to 1 pound AI/A provided excellent control for many of our problem weeds but crop injury was sometimes high on very sandy soils. Cultivars showed injury differences for some post-emergence products; Callisto (mesotrione) damaged both Cave-In-Rock and Shawnee varieties more than Kanlow. Clipping weeds (leaving a 6-inch stubble) also reduced shading effects on seedling grasses during the first year. Over-fertilization stimulated weed growth, and was not necessary for switchgrass production.

For biofuel production we harvested 2 times per growing season beginning the year after planting, though it takes three to five years to develop mature plants and maximum yields. The first harvest occurred in early to mid-July and second at the end of the season in late September or early October. In our experience, adequate stubble height (5-6 inches) is essential to sustain the crop and prevent winter damage, but otherwise long-term survival should not be an issue as long as good agronomic management is provided.

In the Lower Yakima Valley and Columbia Basin switchgrass broke dormancy from early to mid-April but had less than six inches of growth by May 1st. Early season growth is dependent upon irrigation and temperature in our region, as well as variety. Early switchgrass varieties will be 20 inches or taller by late May. With increasing June summer temperatures, growth increases significantly (Figure 22.B2).

12 months May, 2005

13 months, June 2005

First Harvest: June 24, 2005

24 months, May 2006

Figure 22.B2. Switchgrass maturity.

July growth and re-growth was rapid as long as soil moisture was maintained. The first harvest (during the first half of July) ranged from about 4 feet to more than 6 feet of plant matter (Figure 22.B3). Re-growth was observed within 5 - 7 days but sometimes took as long as 10 days. Growth slowed during August compared to July, possibly due to reduced photoperiod.

Figure 22.B3. Switchgrass harvest July, 2006

By September, growth was much slower, and plants entered deep dormancy in late October or early November. No winterkill was seen with any switchgrass varieties, probably due to good irrigation management and cutting regime. Even during December 2003, when record low temperatures occurred (-19^{o}F) while the first switchgrass planting was in the juvenile stage, all varieties survived without winterkill problems.

Our trials have included cultivars with a variety of maturity dates. Dacotah, an upland cultivar, is the earliest maturing switchgrass in our variety trials. It was fully headed by July 1st, several weeks before other varieties and may be too early for biofuel production in the lower Columbia Basin region. Instead, we believe this variety may be best adapted to a higher elevation, shorter growing season where natural precipitation is adequate for the deeply rooted plant to survive.

Kanlow, a lowland variety and late cultivar, has performed very well at both locations, and is the most promising cultivar in our trials. Alamo, a very late maturing cultivar, has very weak stand development with an open canopy. This has allowed for greater weed invasion than any other variety in our research.

Other varieties evaluated include Cave-In-Rock, Trailblazer, Blackwell, Nebraska 28, Sunburst, Forestburg and Shawnee.

Appendix 22C. Additional Data and Explanation of C_4 Fingerprinting for Determining C Sequestration in Switchgrass.

H. Collins and S. Fransen

Additional Data

Table 22.C1. Average physical and chemical properties of the Quincy fine sand (Xeric Torripsamments) soil under switchgrass at the USDA-ARS Integrated Agricultural Research Field Station, Paterson, WA. Average of the three cultivars.

Depth	BD†	Sand	Silt	Clay	pH	Total C	Organic C‡	Total N	C:N
cm	Mg m^{-3}	---------g kg^{-1}--------				-----------g kg^{-1}-----------			
0-15	1.33	917	56	27	6.6	3.6	3.6	0.37	9.5
15-30	1.54	927	52	21	6.3	1.5	1.5	0.22	7.3
30-45	1.61	936	48	16	6.4	1.2	1.2	0.15	8.0
45-60	1.60	928	48	24	7.4	1.6	0.9	0.12	7.5
60-75	1.58	948	38	12	8.1	2.9	0.9	0.10	9.0
75-90	1.60	978	14	8	8.1	3.4	0.8	0.10	8.0

†BD- soil bulk density. ‡Carbonates removed.

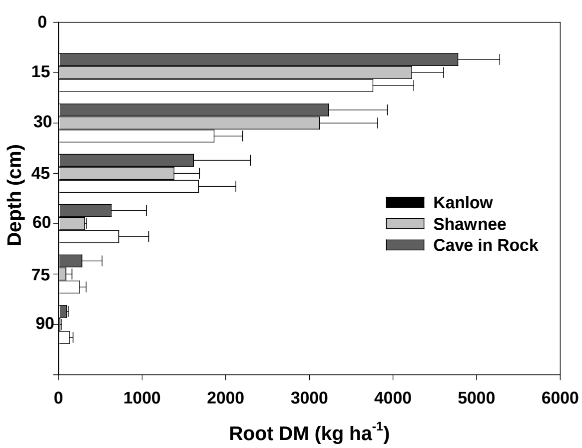


Figure 22.C1. Root biomass (DM – dry matter) in increments to 0.9 m depth for Kanlow, Shawnee and Cave in Rock cultivars sampled in 2006. Root biomass was plotted at the bottom of each sampling increment. Horizontal bars are standard errors of the mean.

Table 22.C2. Soil organic C, δ ^{13}C and percent of C derived from C_4-Cplant growth.

Cultivar	Depth	Organic C		δ ^{13}C	‡Soil C_4-C	C derived from C_4
	cm	mg/kg	kg ha^{-1}	‰	%	kg ha^{-1}
Native soil	0-15	2 992 a *	5 969	-24.9 a *	----	----
	15-30	1 383 b	3 196	-24.0 a *	----	----
	30-45	1 090 bc	2 777	-23.3 b *	----	----
	45-60	990 c	2 376	-23.3 b *	----	----
	60-75	886 c	2 099	-22.9 c	----	----
	75-90	857 c	2 057	-22.7 c	----	----
	Total	----	**18 474**	----	----	----
Cultivated†	0-20	3 180	8 460	-23.9 b	8.2	695
Kanlow	0-15	3 597 a	7 175	-21.9 a	22.9 a	1 645
	15-30	1 527 b	3 527	-22.0 a	20.6 a	727
	30-45	1 173 c	2 834	-21.7 a	14.3 b	405
	45-60	920 d	2 207	-21.7 a	6.7 c	148
	60-75	908 d	2 152	-21.9 a	9.5 c	204
	75-90	864 d	2 074	-22.2 a	5.0 c	104
	Total	----	**19 970**	----	----	**3 233**
Shawnee	0-15	3 627 a	7 235	-21.8 a	23.6 a	1 708
	15-30	1 597 b	3 688	-21.6 a	19.9 a	734
	30-45	1 267 bc	3 059	-21.6 a	13.4 b	410
	45-60	1 015 c	2 436	-21.9 a	5.1 c	124
	60-75	887 cd	2 102	-21.9 a	9.4 c	197
	75-90	781 d	1 874	-21.9 a	7.3 c	137
	Total	----	**20 394**	----	----	**3 310**
Cave in Rock	0-15	3 608 a	7 195	-21.3 a	24.2 a	1 741
	15-30	1 530 b	3 534	-21.0 a	23.9 a	844
	30-45	1 207 bc	2 914	-21.6 a	14.9 b	434
	45-60	1 108 c	2 660	-22.0 a	9.5 c	253
	60-75	870 d	2 061	-22.1 a	7.3 c	151
	75-90	820 d	1 967	-21.7 a	9.6 c	189
	Total	----	**20 331**	----	----	**3 612**

†Cultivated – prior to switchgrass monoculture. ‡The fraction of soil organic-C resulting from the conversion to a monoculture of C_4 switchgrass. % = (δ ^{13}C cropped soil - δ ^{13}C native soil)/(δ ^{13}C switchgrass - δ ^{13}C native soil).Values followed by the same letter within a column and treatment are not significantly different at P=0.05. * Denotes significant difference between native and cultivated soil.

C_4-Derived Carbon, Detailed Description of Methodology and Analysis

Switchgrass uses a process known as C_4 fixation to "fix" atmospheric carbon dioxide for sugar production through photosynthesis. The C_4 process is an elaboration of C_3 carbon fixation that operates in most plants, and is used by approximately 1% of the earth's known plant species. (The name "C_4" comes from the fact that the first product of CO_2 fixation in these plants has four carbon atoms, rather than three, as is the case in C_3 plants.)

The change from C_3 (native shrub-steppe) to cultivation and incorporation of C_4 plants (corn, switchgrass) resulted in a significant change in the $\delta\ ^{13}C$ content of the soil and provided an indicator to determine C sequestration and rates of C turnover. ^{13}C is incorporated into plants by photosynthesis and eventually into soils through decomposition processes. The isotope concentration of a plant is retained essentially unchanged in the SOM (Boutton 1996). Switchgrass is a C_4 plant with a $\delta^{13}C$ -12‰ and when incorporated into a C_3 dominated soil environment (-25‰) allows for accurate assessments of the change in C dynamics (Garten and Wullschleger, 2000).

We calculated the change in the fraction of C_4-C resulting from conversion to a monoculture of C_4 switchgrass using the equation:

$$\%\text{ of C derived from switchgrass} = \frac{\delta\ ^{13}C\text{ cropped soil} - \delta\ ^{13}C\text{ native soil}}{\delta\ ^{13}C\text{ switchgrass} - \delta\ ^{13}C\text{ native soil}}$$

This equation provides the percentage of C in the cropped soil derived from C_4 plants. We then used an exponential decay equation to determine the turnover rate of the C_3-C within the 0-15 and 15-30 cm depth increments; $A_t = A_o(e^{-kt})$, where, A_o is the amount of C_3-C in the soil prior to the C_4 monoculture, A_t is the amount of C_3-C remaining after replacement by C_4-C vegetation (corn and switchgrass), k is the decay constant and t was the time of cropping (5 y). We assumed that the $\delta\ ^{13}C$ of cropped soil samples prior to cropping were similar to that of the adjacent native shrub-steppe site.

For a full description of methods, see Collins et al. *in press.*

The $\delta\ ^{13}C$ profile of the non-cultivated shrub steppe soil differed significantly from the soil cropped to switchgrass monocultures (Table 22.C2, above). The $\delta^{13}C$ of the native shrub steppe vegetation was -24.9‰, with the uncultivated soil averaging -24.5‰ in the 0-15 and 15-30 cm depth increments and -23.1‰ to 90 cm (Table 22.C2). Soils collected prior to establishment of the switchgrass monocultures showed a $\delta\ ^{13}C$ of -23.9 ‰, a 1 ‰ enrichment over the native soil, reflecting the incorporation of corn residues from the 2002 crop year. The $\delta\ ^{13}C$ of the switchgrass aboveground biomass and roots averaged -12‰ among cultivars, with soil cropped to switchgrass -21.9‰ to 90 cm. The surface horizon of soil cropped to Kanlow and Shawnee had a $\delta\ ^{13}C$ enrichment of 3‰ above the native uncultivated soil, and 3.6‰ for Cave in Rock. The higher soil enrichment of ^{13}C under Cave in Rock reflects the greater concentration of roots in the 0-15 and 15-30 cm depth increments compared to soils cropped to Kanlow and Shawnee (Figure 22.C1, above). Enrichment in the 30-60 cm [30-45 and 45-60 cm] and 60-90 cm [60-75 and 75-90 cm] depth increments averaged 1.7‰ and 0.9‰, respectively.

From these soil enrichment analyses we calculated the change in the fraction of SOC resulting from native shrub-steppe conversion to monocultures of switchgrass (Table 22.C2). On average 23.6% of the SOC (580 mg C kg^{-1} soil or 1.7 Mg C ha^{-1}) present in the 0-15 cm increment was derived from C_4 cropping (corn and switchgrass) since conversion. The percentage of C_4-C in the SOC averaged 21.5% in the 15-30 cm, 14.2% in the 30-45 cm depth increments and 7% below 45cm, reflecting the decline in switchgrass root biomass. Total soil profile C_4-C to 90 cm was 3.2, 3.3 and 3.6 Mg C ha^{-1} for soil cropped to the Kanlow, Shawnee and Cave in Rock cultivars, respectively.

As stated in the site history, the field had been cropped to one year of corn (C_4, -12‰) in 2002 and potato (C_3, - 25‰) in 2003. Soils collected prior to establishment of the switchgrass monocultures showed a δ ^{13}C soil enrichment of 1.0‰ indicating 8.2 % of the SOC was derived from the previous corn crop Field corn biomass incorporated in 2002 input 4.6 Mg C_4-C ha^{-1}. We calculated based on the ^{13}C enrichment prior to the switchgrass monoculture that 695 kg C_4-corn C ha^{-1} was incorporated into the soil organic matter in the surface 20 cm. This shows that the attempt to use a mass C balance as described in the previous section underestimated the C contribution from the single corn crop by nearly 35 % Because potato is a C_3 plant we were unable to determine the contribution of potato-C to the total C pool from the background of the native vegetation C. In the previous section we estimated that 100 kg potato C ha^{-1} was incorporated into various soil organic matter pools in the surface 30 cm, this value also most likely underestimates the contribution of the C3 potato inputs. However, contributions from C_4 sources were easily distinguished and pool sizes could be estimated.

On average 23.6% of the SOC (580 mg C kg^{-1} soil or 1.7 Mg C ha^{-1}) present in the 0-15 cm increment was derived from C_4 cropping (corn/switchgrass) since conversion. The percentage of C_4-C in the SOC averaged 21.5% in the 15-30 cm, 14.2% in the 30-45 cm depth increments and 7% below 45cm, reflecting the decline in switchgrass root biomass. Total soil profile C_4-C to 90 cm was 3.2, 3.3 and 3.6 Mg C ha^{-1} for soil cropped to the Kanlow, Shawnee and Cave in Rock cultivars, respectively. Subtracting the estimate of C_4-C (695 kg C ha^{-1}) derived from the addition of corn residues in 2002, we conclude that ~1800 kg C ha^{-1} (73 %) of the total C_4-C present within the surface 30 cm was derived from switchgrass during three years of switchgrass cropping. Roots were the primary source of the new SOC since above ground biomass was removed at each harvest. The amount of soil profile C_4-C determined by $\delta^{13}C$ analysis showed a greater input of C (average, 3368 kg C_4 ha^{-1}) than that determined by the difference in total profile C (1758 kg C ha^{-1}) between the uncultivated native soil and soils cropped to switchgrass described previously. Over the three years of cropping ~1600 kg of C_3-C ha^{-1} was replaced by C_4-C in addition to the profile C increase (1758 kg C ha^{-1}) that was not picked up by the analysis of the change in mass of soil C. The average accrual rate of C_4-SOC was estimated at 1.1 Mg ha^{-1} y^{-1}. This accrual rate was twice that of sequestration estimated by just the change in total C and is in line with the results of other research describing C sequestration by switchgrass (Liebig et al., 2008; Garten and Wullschleger, 2000; McLaughlin and Walsh, 1998). Garten and Wullschleger (2000) using δ ^{13}C analysis found that 19 to 31% of the SOC had been derived from switchgrass roots. Even with aboveground biomass removal root contributions to

SOC were positive. It is clear that the change in δ ^{13}C of the cropped soil reflects the shift to C_4-C inputs from switchgrass.

Soil Organic Carbon and Turnover Time

The CO_2 evolved during C mineralization was used to determine the size and kinetics of the functional C pools of soil (Paul et al, 1999; Collins et al., 2000) for each depth increment for each monoculture of switchgrass. The size and turnover rates of each pool were estimated by curve fitting the CO_2 evolved per unit of time (C_t) using a three-component first-order model:

$$C_t = C_a e^{-kat} + C_s e^{-kst} + C_r e^{-krt}$$

where; C_a = size of the active C pool in g C kg^{-1}, k_a = is the C_a mineralization rate; C_s = size of the slow C pool in g C kg^{-1}, k_s = is the C_s mineralization rate; and C_r = size of the resistant C pool in g C kg^{-1}, k_r = is the C_r mineralization rate. Three parameters, C_a, k_a, and k_s were estimated using the non-linear regression model (NonLIN) of Systat (Systat, Inc., Evanston, IL). Since the residue of acid hydrolysis typically carbon date greater than 500 y it was assumed that negligible amounts of the CO_2 evolved during the extended incubation were derived from the C_r pool (Paul et al., 1997). This assumption made it possible to analyze the CO_2 data as the sum of two first order rate reactions. The slow pool C_s pool was defined as $C_s = C_t - C_a - C_r$. The model was based on the assumption of first-order kinetics, i.e., where the rate of C mineralization is proportional to the amount of C in the organic matter pool. When integrated over time this produces an exponential decay curve. MRT was reported as the reciprocal of the decomposition rate constant (k^{-1}) derived from the first order rate reactions. Acid hydrolysis determined the size of the resistant C pool (C_r). The acid resistant organic fraction was determined by digesting 1 g soil in 6 N HCl for 18 h at 120 ^oC. Digested samples were washed three times with deionized water, dried at 55ºC, and ground to pass a 180 µm screen. Results are presented for only the non-hydrolyzable fraction. The acid soluble fraction can be estimated by difference. Non-hydrolyzable C was determined by dry combustion on a LECO, CNS-2000 Elemental Analyzer St. Joseph, MI. See Collins et al., *in press* for a full description of methods.

Pool sizes and C-mineralization kinetics for the active (C_a), slow (C_s) and resistant (C_r) C pools are presented in Table 22.C3. Figure 22.C2 shows the rates of C-mineralized during extended laboratory incubations for the 0-15, 15-30, 30-60 and 60-90 cm depth increments of the uncultivated soil and soils cropped to the switchgrass cultivars. Switchgrass cultivar did not significantly affect the size or turnover of C in any pool. Cumulative C mineralized during the 375 d laboratory incubation was not significantly different among the uncultivated native soil and switchgrass monocultures but was significant with depth, averaging 750, 353, 275, and 195 mg CO_2-C kg^{-1} for the 0-15, 15-30, 30-60 and 60-90 cm depth increments, respectively. The percentage of the total C-mineralized within a depth increment during the incubation increased with depth, ranging from 21 – 27%.

Table 22.C3. Pool sizes and C-mineralization kinetics of soil for the active and slow C pools from the 0-15, 15-30, 30-60 and 60-90 cm depth increments for soils cropped to switchgrass cultivars and the native vegetation at the USDA-ARS, Integrated Cropping Systems Research Field Station located near Paterson, Benton County, Washington.

		C-Mineralization		Active Pool		Slow Pool		Resistant Pool	
Cultivar	Depth	Cum. CO_2-C	C-min /SOC	C_a	MRT	C_s	MRT	C_r	C_r /SOC
	cm	mg kg^{-1}	(%)	mg kg^{-1}	-- d --	mg kg^{-1}	--- y ---	mg kg^{-1}	%
Native	0-15	781 a	26.1	373 a *	66 a *	1 611 a*	4.2 a *	1 008 a *	34 a *
	15-30	357 b	25.8	107 b *	46 b *	640 b	2.3 b	636 b	46 b
	30-60	261 c	25.1	22 c	19 c	405 b	1.5 b	612 b	57 b
	60-90	237 c	27.2	30 c	27 c	458 b	2.0 b	458 c	46 b
Kanlow	0-15	776 a	21.6	179 a	29 a	1874 a	2.8 a	1 545 a	43 a
	15-30	376 b	24.6	52 b	22 ab	817 b	2.2 a	655 b	43 a
	30-60	289 c	27.6	30 c	26 a	393 c	1.3 b	624 b	54 a
	60-90	180 d	20.3	17 c	16 b	472 c	2.4 a	397 c	43 a
Shawnee	0-15	748 a	20.6	187 a	39 a	1819 a	2.8 a	1 622 a	45 a
	15-30	348 b	21.8	89 b *	22 b	650 b	2.0 b	859 b	54 a
	30-60	291 c	25.5	52 c	25 b	508 bc	1.6 b	646 bc	51 a
	60-90	164 d	19.7	23 d	27 b	403 c	2.3 ab	474 c	46 a
Cave in Rock	0-15	692 a	19.2	151 a	31 a	1951 a	3.3 a	1 506 a	42 a
	15-30	331 b	21.6	62 b	18 b	557 b	1.7 b	911 b	54 a
	30-60	262 c	22.6	35 c	20 b	478 bc	1.7 b	646 c	48 a
	60-90	200 c	23.5	22 c	18 b	351 c	1.7 b	449 c	40 a

§Values followed by the same letter within a column by treatment are not significantly different at P=0.05. Values between treatments followed by "*" within a depth increment are significantly different at P=0.05.

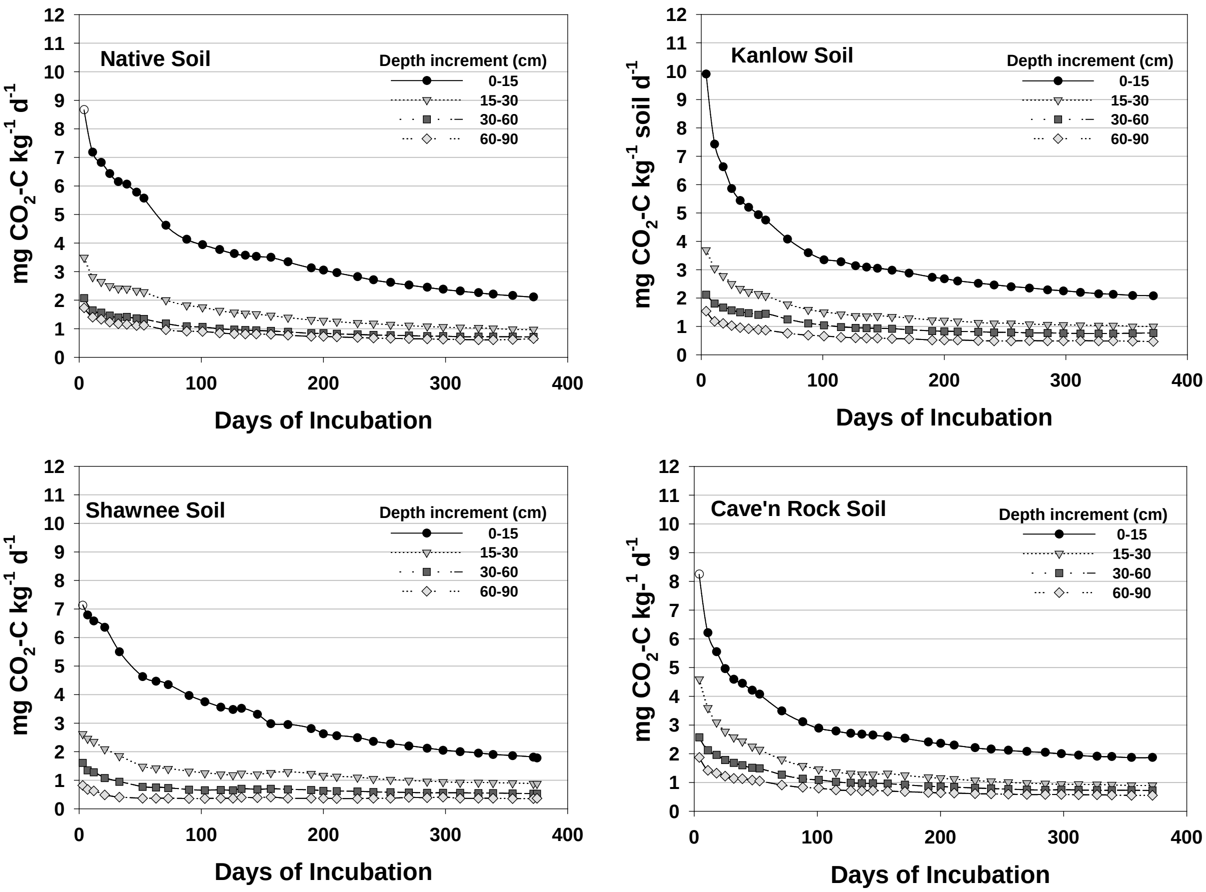


Figure 22.C2. Rate of C-mineralization for soils obtained from the native shrub steppe, Kanlow Shawnee, and Cave in Rock switchgrass cultivar plots.

Soil organic C has been routinely divided into three functional pools: an active pool (C_a) consisting of labile C with a mean residence time (MRT= 1/k) of days; an intermediate or slow pool (C_s) consisting of structural plant residues and physically stabilized C with an MRT of 25-50 y; and a resistant fraction (C_r) consisting of chemically stabilized C with an MRT of 1000-1500 yr (Paul et al., 2006; Paul et al., 2001; Collins et al., 1999; 2000; Buyanovsky et al., 1994). Our laboratory has routinely used extended laboratory incubations of soil with measurements of CO_2 to differentiate the C_a and C_s functional C pools in residues and derive estimates of SOC turnover (Collins et al., 1999; Paul et al., 1999; Collins et al., 2000; Paul et al., 2001b; Cochran et al., 2007; Haile-Mariam et al., 2008). This method can be considered a biological fractionation of organic matter, whereby the labile (active) fractions (C_a) of SOM are rapidly mineralized by soil microorganisms and subsequent soil C (C_s) fractions are more slowly mineralized (Paul et al., 1999).

The active (C_a) pool of C in the 0-15 cm depth increment of the uncultivated soil comprised 12.5% of the total SOC and was significantly greater than the active pools of the cropped switchgrass monocultures, where the C_a pool comprised 5% of the total soil C. The size of the C_a pool declined significantly with depth (Table 22.C3, above). The larger C_a pool of the uncultivated soil compared to the irrigated switchgrass monocultures reflects the low precipitation conditions of the shrub-steppe climate (~178 mm y^{-1}) that would limit decomposition, thus building up labile C. Upon cultivation and application of irrigation water the C_a pool rapidly decomposed as evidenced by the smaller C_a pool of the switchgrass monocultures. Within switchgrass soil profiles, 57% of the active pool was present in the 0-15 cm depth increment reflecting the greater concentration of roots in that horizon. The laboratory C_a-MRT of the uncultivated soil was 66 d in the 0-15 cm declining to an average of 27 d to 90 cm, where C_a-MRTs for the switchgrass monocultures averaged 24 d throughout the profile. The short MRTs of the C_a pool reflect the rapid turnover of labile C originating from switchgrass roots and exudates (Johnson et al., 2007).

The proportion of total C in the slow pool (C_s) averaged 54% in the surface 15 cm among the switchgrass monocultures and uncultivated soil and decreased to an average of 44% below 15 cm. The C_s was significantly greater only in the 0-15 cm depth increment of the switchgrass monocultures (Avg. = 1823 mg kg^{-1}) compared to the uncultivated (1611 mg kg^{-1}) soil. The size of the C_s pool within each depth increment below 15 cm was not significantly different among switchgrass monocultures and the uncultivated soil. The laboratory C_s-MRT of the uncultivated soil was 4.5 y in the 0-15 cm declining to an average of 1.9 y to 90 cm. Where, C_s-MRTs for the switchgrass monocultures where significantly lower averaging 2.8 y for Kanlow and Shawnee and 3.3 y for Cave in Rock in the 0-15 cm and 1.9 y to 90 cm, similar to the uncultivated soil. The slow SOM fraction is made up of lignin derived plant material and stabilized microbial products (Paul et al., 2006). This fraction makes up approximately 55% of the total SOM.

The C_r-C fraction accounted for 34% of the total C in the 0-15 cm depth increment of the uncultivated native soil and 43% for the switchgrass monocultures (Table 22.C3). The C_r-C fraction generally makes up 40- 50% of the total SOM and will have a higher value for clay soils (Paul et al., 2006; Paul et al., 2001; Collins et al., 2000). The low concentration of C_r-C in the uncultivated soil reflects the arid conditions of the region that limit decomposition and stabilization of C. Also there would be little physical protection through aggregation due to the single grain nature of this 95% sand soil. Upon irrigation in 2002 combined with high soil temperatures of the region the Cr-C pool increased as decomposition of labile plant residues and mineralization of the C_a pool of soil organic matter accelerated. Table 22.C3 shows a ~50% loss in the Ca pool which would redistribute the proportion of C within each pool.

The difference in assimilation of ^{13}C by C_3 and C_4 plants provided a useful approach to assess C turnover (Collins et al., 1999; Follett et al., 1997; Gregorich et al., 1995). Assuming an exponential decay of the SOC, mean residence times of C_3-C remaining after 5 y since conversion in the 0-15 depth increment of the cultivated soils were 66, 65 and 56 y, for the Kanlow, Shawnee and Cave in Rock monocultures, respectively (Table 22.C4). For the 15-30 cm increment MRTs were 50, 55 and 30 y, respectively. This analysis reflected the higher accumulation of SOC in the surface than at depth in arid environments. In subsoil horizons, the decomposable pool consisted mainly of root biomass and from labile soluble fractions of C transported from overlying horizons accumulating with depth.

Table 22.C4. Estimates of decomposition rates for C_3-C soil organic matter based on shifts in the natural abundance of $\delta^{13}C$ in soil for the 0-15 and 15-30 cm depth increments now under C_4-switchgrass.

	Native	Kanlow			Shawnee			Cave in Rock		
Depth	A_o†	A_t	-k	C_3-C MRT	A_t	-k	C_3-C MRT	A_t	-k	C_3-C MRT
cm	---- mg C kg^{-1} ----			y	mg C kg^{-1}		y	mg C kg^{-1}		y
0-15	2 992	2 773	-0.0152	66	2 771	-0.0154	65	2 735	-0.0180	56
15-30	1 383	1 252	-0.0199	50	1 263	-0.0182	55	1 167	-0.0283	35

†$A_t = A_o(e^{-kt})$, where A_o is the total C_3-C prior to conversion to C_4 switchgrass, A_t is the proportion of C_3-C remaining at the time (t) of sampling, k is the decay constant and MRT is the mean residence time (1/k0).

Appendix 22D. Additional Data from Biochar Characterization, and Soil and Plant Effects.

H. Collins, J. Streuble, and D. Granatstein

Table 22.D1. C, N, and S concentrations of biochar feedstock

Feedstock	C	N	S	C:N	C:S
	--------------------g kg-1--------------------				
Switchgrass	432 (8)*	23.5 (0.3)	1.2 (0.2)	18	360
Digested Fiber	480 (2)	20.0 (0.2)	3.3 (0.2)	24	145
Softwood Bark	470 (2)	3.3 (0.1)	0.3 (0.1)	142	1567
Wood Pellets	477 (9)	1.2 (0.1)	0.3 (0.1)	398	1590

*Std. error of mean in parentheses

Table 22.D2. Selected characteristics of the five soil types used in the laboratory analysis.

		Soil Characteristics						
Soil Series	Texture	C	N	S	C:N	C:S	pH	CEC
		-------------g/kg-------------						cmol/kg
Quincy	sand	4.3 (0.5)†	0.5 (0.1)	0.2 (0.06)	8.6	22	7.1	3.3
Naff	silt loam	18.5 (0.4)	1.5 (0.1)	0.1 (0.03)	12.3	185	4.5	15.4
Palouse	silt loam	23.3 (1.1)	1.9 (0.1)	0.2 (0.01)	12.3	117	4.6	16.0
Thatuna	silt loam	27.5 (1.1)	2.3 (0.1)	0.3(0.02)	12.0	92	4.6	16.1
Hale	silt loam	39.4 (0.2)	3.3 (0.1)	0.6 (0.05)	12.0	66	4.6	16.6

†Standard error of mean in parentheses.

Table 22.D3. Concentrations of soil C and N, pH , CEC and water holding capacity after additions of biochars (500 °C) to the Quincy sand soil.

Soil		Biochar	Soil + Biochar								
Series	Biochar	†Rate	C	N	S	C:N	C:S	pH	CEC	Water Holding (%)	
		Mg ha^{-1}	------- g kg^{-1} soil------						cmol kg^{-1}	0 MPa§	0.1 MPa
Quincy	Switchgrass	0	4.3 (0.5) ‡ a	0.5 (0.05) a	0.2 (0.03) a	9	22	7.1 a	3.3	26.0 (0.3) a	4.0 (0.1) a
		9.8	2.6 (0.1) b	0.2 (0.03) b	0.2 (0.08) a	13	13	7.8 b	Nd	29.3 (0.9) b	4.1 (0.1) ab
		19.5	4.6 (0.3) ac	0.2 (0.01) b	0.2 (0.04) a	23	23	7.9 b	Nd	29.8 (0.5) b	4.3 (0.1) bc
		39.0	8.9 (0.3) d	0.3 (0.04) c	0.1 (0.01) b	30	89	7.9 b	Nd	31.3 (1.0) c	4.5 (0.2) c
	Digested	0	4.3 (0.5) a	0.5 (0.05) a	0.2 (0.03) a	9	22	7.1 a	3.3	26.0 (0.3) a	4.0 (0.1) a
	Fiber	9.8	2.5 (0.6) b	0.2 (0.05) b	0.1 (0.04) b	13	25	7.8 b	4.2	27.5 (0.9) b	4.5 (0.2) b
		19.5	4.9 (0.3) ac	0.2 (0.01) b	0.2 (0.04) a	25	25	8.0 c	4.2	29.4 (0.5) c	4.5 (0.2) b
		39.0	9.9 (0.7) d	0.4 (0.04) c	0.2 (0.02) a	25	50	8.1 c	4.4	32.5 (1.0) d	4.4 (0.3) b

Softwood	0	4.3 (0.5) a	0.5 (0.05) a	0.2 (0.03) a	9	22	7.1 a	3.3	26.0 (0.3) a	4.0 (0.1) a
Bark	9.8	3.3 (0.3) b	0.1 (0.02) b	0.1 (0.05) b	33	33	7.4 b	4.3	26.1 (1.2) a	3.9 (0.2) a
	19.5	5.7 (0.7) ac	0.1 (0.02) b	0.1 (0.04) b	57	57	7.9 c	4.4	27.8 (0.3) b	4.0 (0.2) a
	39.0	9.4 (0.4) d	0.2 (0.06) c	0.1 (0.12) b	47	94	8.1 c	4.4	29.0 (0.8) c	4.0 (0.1) a
Wood	0	4.3 (0.5) a	0.5 (0.05) a	0.2 (0.03) a	9	22	7.1 a	3.3	26.0 (0.3) a	4.0 (0.1) a
Pellets	9.8	4.5 (0.7) ab	0.1 (0.03) b	0.2 (0.08) a	45	47	7.0 a	4.1	26.1 (0.9) a	4.1 (0.1) ab
	19.5	5.6 (0.7) b	0.1 (0.02) b	0.2 (0.11) a	56	16	7.6 b	4.2	26.3 (0.6) a	4.2 (0.1) b
	39.0	11.2 (1.4) c	0.1 (0.01) b	0.1 (0.06) a	112	35	7.6 b	4.2	26.0 (1.5) a	4.0 (0.1) a

†Rate of biochar application. Mg/ha=megagrams per hectare=metric tons per hectare. 1 metric ton per hectare = 890 lb/acre or 0.445 short tons per acre. A short ton = 2000 lb. Nd –not determined at the time of this report. ‡Std. error of mean in parentheses. Statistical comparisons were not made among biochars. Values for a biochar within a column followed by the same letter are not significantly different at p = 0.05. §MPa = 0.1 bar.

Table 22.D4. Concentrations of soil C and N, pH, CEC and water holding capacity after additions of biochars (500°C) to the Naff silt loam soil type.

Soil		Biochar	Soil + Biochar								
Series	Biochar	†Rate	C	N	S	C:N	C:S	pH	CEC	Water Holding (%)	
		Mg ha^{-1}	-------- g kg^{-1}-------						cmol kg^{-1}	0 MPa§	0.1 MPa
Naff	Switchgrass	0	18.0 (1.0) ‡ a	1.5 (0.05) a	0.2 (0.02) a	12	90	4.5 a	15.4	50.3 (3.1) a	Nd
		9.8	19.9 (0.4) b	1.6 (0.03) b	0.2 (0.01) a	12	100	4.7 b	Nd	52.7 (1.6) a	Nd
		19.5	22.6 (0.9) c	1.7 (0.04) c	0.2 (0.02) a	13	113	4.9 c	Nd	49.4 (1.4) a	Nd
		39.0	27.8 (0.6) d	1.8 (0.04) d	0.2 (0.01) a	15	139	5.0 c	Nd	49.6 (1.3) a	Nd
	Digested	0	18.0 (1.0) a	1.5 (0.05) a	0.2 (0.02) a	12	90	4.5 a	15.4	50.3 (3.1) a	Nd
	Fiber	9.8	20.7 (1.0) b	1.6 (0.05) b	0.2 (0.03) a	13	104	4.7 b	16.1	52.7 (2.0) a	Nd
		19.5	22.9 (0.6) c	1.7 (0.02) c	0.2 (0.04) a	14	115	4.8 b	16.6	51.2 (1.9) a	Nd
		39.0	26.6 (0.4) d	1.8 (0.03) d	0.2 (0.04) a	15	133	5.3 c	16.8	53.5 (2.3) a	Nd

Softwood	0	18.0 (1.0) a	1.5 (0.05) a	0.2 (0.02) a	12	90	4.5 a	15.4	50.3 (3.1) a	Nd
bark	9.8	21.4 (0.3) b	1.5 (0.01) a	0.2 (0.03) a	14	107	4.8 b	17.1	54.1 (5.1) a	Nd
	19.5	23.0 (0.7) c	1.5 (0.03) a	0.2 (0.01) a	15	115	4.8 b	17.2	52.3 (5.9) a	Nd
	39.0	30.2 (0.7) d	1.5 (0.03) a	0.2 (0.01) a	20	151	4.9 c	18.6	50.9 (2.3) a	Nd
Wood	0	18.0 (1.0) a	1.5 (0.05) a	0.2 (0.02) a	12	90	4.5 a	15.4	50.3 (3.1) a	Nd
Pellets	9.8	20.8 (0.9) b	1.4 (0.04) a	0.2 (0.01) a	15	104	4.6 b	15.5	51.1 (0.7) a	Nd
	19.5	24.3 (1.8) c	1.5 (0.03) a	0.3 (0.05) b	16	81	4.6 b	15.8	50.6 (2.3) a	Nd
	39.0	34.2 (1.1) d	1.4 (0.06) a	0.3 (0.04) b	24	114	4.8 c	16.1	57.2 (2.4) a	Nd

†Rate of biochar application. Mg/ha=megagrams per hectare=metric tons per hectare. 1 metric ton per hectare = 890 lb/acre or 0.445 short tons per acre. A short ton = 2000 lb. Nd –not determined at the time of this report. ‡Std. error of mean in parentheses. Statistical comparisons were not made among biochars. Values for a biochar within a column followed by the same letter are not significantly different at $p = 0.05$. §MPa = 0.1 bar.

Table 22.D5. Concentrations of soil C and N, pH, CEC and water holding capacity after additions of biochars (500°C) to the Palouse silt loam soil type.

Soil		Biochar	Soil + Biochar								
Series	Biochar	†Rate	C	N	S	C:N	C:S	pH	CEC	Water Holding (%)	
		Mg ha^{-1}	-------- g kg^{-1}-------						cmol kg^{-1}	0 MPa§	0.1 MPa
Palouse	Switchgrass	0	23.2 (0.5) ‡ a	2.0 (0.04) a	0.4 (0.11) a	12	58	4.6 a	16.0	53.9 (3.4) a	Nd
		9.8	26.0 (0.3) b	2.0 (0.01) a	0.3 (0.04) a	13	87	4.7 a	Nd	53.3 (2.2) a	Nd
		19.5	28.3 (0.7) c	2.1 (0.04) b	0.4 (0.13) a	13	71	4.9 b	Nd	55.2 (3.9) a	Nd
		39.0	32.0 (0.6) d	2.2 (0.01) c	0.3 (0.04) a	15	107	5.1 c	Nd	55.5 (2.7) a	Nd
	Digested	0	23.2 (0.5) a	2.0 (0.04) a	0.4 (0.11) a	12	58	4.6 a	16.0	53.9 (3.4) a	Nd
	Fiber	9.8	25.6 (0.3) b	2.0 (0.04) a	0.5 (0.13) a	13	51	4.9 a	16.0	51.8 (1.4) a	Nd
		19.5	29.4 (0.7) c	2.2 (0.04) b	0.4 (0.05) a	13	74	4.9 b	16.3	57.9 (4.6) a	Nd
		39.0	38.1 (0.6) d	2.6 (0.04) c	0.4 (0.03) a	15	95	5.3 c	16.6	52.6 (4.3) a	Nd

Softwood	0	23.2 (0.5) a	2.0 (0.04) a	0.4 (0.11) a	12	58	4.6 a	16.0	53.9 (3.4) a	Nd
bark	9.8	27.2 (1.0) b	2.2 (0.05) a	0.3 (0.05) a	12	91	4.8 a	16.1	55.5 (2.3) a	Nd
	19.5	28.2 (0.7) c	2.3 (0.07) a	0.3 (0.03) a	12	94	4.8 a	16.2	61.4 (1.5) b	Nd
	39.0	36.5 (0.8) d	2.2 (0.01) a	0.4 (0.12) a	17	91	4.9 b	17.6	60.3 (0.7) b	Nd
Wood	0	23.2 (0.5) a	2.0 (0.04) a	0.4 (0.11) a	12	58	4.6 a	16.0	53.9 (3.4) a	Nd
Pellets	9.8	26.3 (0.6) b	1.9 (0.02) a	0.3 (0.04) a	14	88	4.6 a	15.8	58.5 (4.0) a	Nd
	19.5	34.9 (1.3) c	2.2 (0.04) a	0.3 (0.07) a	16	116	4.6 a	17.5	56.2 (2.2) a	Nd
	39.0	38.7 (1.0) d	2.3 (0.11) a	0.3 (0.05) a	17	129	4.8 b	18.7	53.3 (3.3) a	Nd

†Rate of biochar application. Mg/ha=megagrams per hectare=metric tons per hectare. 1 metric ton per hectare = 890 lb/acre or 0.445 short tons per acre. A short ton = 2000 lb. Nd –not determined at the time of this report. ‡Std. error of mean in parentheses. Statistical comparisons were not made among biochars. Values for a biochar within a column followed by the same letter are not significantly different at p = 0.05. §MPa = 0.1 bar.

Table 22.D6. Concentrations of soil C and N, pH and CEC after additions of biochars (500°C) to the Thatuna silt loam soil type.

Soil		Biochar	Soil + Biochar								
Series	Biochar	†Rate	C	N	S	C:N	C:S	pH	CEC	Water Holding (%)	
		Mg ha^{-1}	-------- g kg^{-1}-------						cmol kg^{-1}	0 MPa§	0.1 MPa
Thatuna	Switchgrass	0	26.9 (0.5) ‡ a	2.4 (0.04) a	0.4 (0.11) a	11	67	4.4 a	16.1	57.2 (1.6) a	Nd
		9.8	28.3 (0.4) b	2.2 (0.03) b	0.3 (0.05) a	13	94	4.5 a	Nd	55.3 (2.0) a	Nd
		19.5	31.0 (0.5) c	2.2 (0.06) b	0.3 (0.07) a	14	103	4.9 b	Nd	59.9 (1.7) a	Nd
		39.0	37.6 (1.5) d	2.4 (0.02) a	0.4 (0.15) a	16	94	5.1 c	Nd	57.7 (2.4) a	Nd
	Digested	0	26.9 (0.5) a	2.4 (0.04) a	0.4 (0.11) a	11	67	4.4 a	16.1	57.2 (1.6) a	18.4 (0.3) a
	Fiber	9.8	29.6 (0.3) b	2.3 (0.05) b	0.3 (0.03) a	13	99	4.6 b	16.1	56.7 (1.0) a	18.5 (0.2) a
		19.5	30.4 (0.3) c	2.3 (0.03) b	0.3 (0.04) a	13	101	4.9 c	17.2	56.8 (2.2) a	18.3 (0.4) a
		39.0	37.2 (0.3) d	2.5 (0.04) a	0.3 (0.04) a	15	124	5.0 c	16.0	56.5 (2.1) a	18.6 (0.1) a

Softwood	0	26.9 (0.5) a	2.4 (0.04) a	0.4 (0.11) a	11	67	4.4 a	16.1	57.2 (1.6) a	Nd
bark	9.8	28.3 (1.1) b	2.1 (0.03) b	0.3 (0.04) a	12	94	4.6 a	16.1	53.9 (1.5) a	Nd
	19.5	31.9 (1.6) c	2.1 (0.01) b	0.3 (0.03) a	12	106	4.8 b	17.8	52.7 (3.4) a	Nd
	39.0	34.9 (2.8) c	2.1 (0.07) b	0.3 (0.03) a	17	116	4.9 b	18.0	58.7 (1.6) a	Nd
Wood	0	26.9 (0.5) a	2.4 (0.04) a	0.4 (0.11) a	11	67	4.4 a	16.1	57.2 (1.6) a	Nd
Pellets	9.8	27.8 (1.3) a	2.1 (0.02) b	0.3 (0.03) a	14	93	4.5 a	15.1	52.9 (2.1) b	Nd
	19.5	30.3 (0.7) b	2.1 (0.07) b	0.3 (0.04) a	16	101	4.6 b	16.4	50.7 (1.9) b	Nd
	39.0	35.9 (2.3) c	2.1 (0.04) b	0.3 (0.04) a	17	120	4.6 b	17.3	56.8 (1.4) a	Nd

†Rate of biochar application. Mg/ha=megagrams per hectare=metric tons per hectare. 1 metric ton per hectare = 890 lb/acre or 0.445 short tons per acre. A short ton = 2000 lb. Nd –not determined at the time of this report. ‡Std. error of mean in parentheses. Statistical comparisons were not made among biochars. Values for a biochar within a column followed by the same letter are not significantly different at $p = 0.05$. §MPa = 0.1 bar.

Table 22.D7. Concentrations of soil C and N, pH, CEC and water holding capacity after additions of biochars (500^{o}C) to the Hale silt loam soil type.

Soil		Biochar	Soil + Biochar								
Series	Biochar	†Rate	C	N	S	C:N	C:S	pH	CEC	Water Holding (%)	
		Mg ha^{-1}	-------- g kg^{-1}-------						cmol kg^{-1}	0 MPa§	0.1 MPa
Hale	Switchgrass	0	39.9 (0.9) ‡ a	3.4 (0.09) a	0.6 (0.10) a	12	67	4.7 a	16.6	52.9 (1.5) a	23.8 (0.9) a
		9.8	43.7 (1.7) b	3.4 (0.13) a	0.6 (0.05) a	13	73	4.7 a	Nd	57.3 (3.1) a	25.0 (0.3) ab
		19.5	44.1 (1.5) b	3.4 (0.11) a	0.6 (0.07) a	13	74	4.9 b	Nd	58.0 (4.1) a	25.0 (0.1) ab
		39.0	49.2 (1.3) c	3.6 (0.11) a	0.6 (0.15) a	14	82	5.0 b	Nd	58.6 (3.0) a	25.4 (0.4) b
	Digested	0	39.9 (0.9) a	3.4 (0.09) a	0.6 (0.10) a	12	67	4.7 a	16.6	52.9 (1.5) a	23.8 (0.9) a
	Fiber	9.8	42.6 (1.4) b	3.4 (0.09) a	0.6 (0.03) a	13	71	4.8 a	15.6	55.3 (2.2) a	25.3 (0.5) b
		19.5	44.7 (1.9) b	3.5 (0.09) a	0.7 (0.04) a	13	64	4.9 b	16.6	61.4 (3.1) a	25.4 (0.4) b
		39.0	48.9 (2.3) c	3.6 (0.14) a	0.7 (0.04) a	14	71	5.1 c	16.5	58.8 (2.0) a	25.3 (0.3) b

Softwood	0	39.9 (0.9) a	3.4 (0.09) a	0.6 (0.10) a	12	67	4.7 a	16.6	52.9 (1.5) a	23.8 (0.9) a
bark	9.8	42.5 (0.5) b	3.4 (0.07) a	0.6 (0.04) a	13	71	4.8 a	15.5	52.6 (4.4) a	24.8 (0.2) a
	19.5	44.5 (1.5) c	3.2 (0.09) a	0.6 (0.03) a	14	74	4.8 a	17.0	55.3 (1.3) a	22.6 (0.3) a
	39.0	49.5 (1.0) d	3.2 (0.13) a	0.6 (0.03) a	15	83	4.9 b	16.7	56.9 (5.5) a	23.1 (0.1) a
Wood	0	39.9 (0.9) a	3.4 (0.09) a	0.6 (0.10) a	12	67	4.7 a	16.6	52.9 (1.5) a	23.8 (0.9) a
Pellets	9.8	43.3 (0.5) b	3.4 (0.12) a	0.6 (0.03) a	13	72	4.7 a	15.7	39.4 (1.9) b	24.8 (0.4) a
	19.5	45.7 (1.3) c	3.2 (0.14) a	0.6 (0.04) a	14	76	4.7 a	17.2	42.4 (2.3) b	24.7 (0.6) a
	39.0	50.3 (2.3) d	3.3 (0.06) a	0.6 (0.04) a	15	84	4.9 b	15.8	39.2 (5.8) b	24.7 (0.2) a

†Rate of biochar application. Mg/ha=megagrams per hectare=metric tons per hectare. 1 metric ton per hectare = 890 lb/acre or 0.445 short tons per acre. A short ton = 2000 lb. Nd –not determined at the time of this report. ‡Std. error of mean in parentheses. Statistical comparisons were not made among biochars. Values for a biochar within a column followed by the same letter are not significantly different at p = 0.05. §MPa = 0.1 bar.

Table 22.D8. Pool sizes and C-mineralization kinetics of soil for the active and slow C pools for the Quincy sand soil amended with 0, 9.8, 19.5 and 39 Mg ha^{-1} biochar.

		C-Mineralization		Active Pool			Slow Pool			Resistant Pool	
Cultivar	Rate	Cum. CO_2-C	C-min /SOC	C_a	Lab MRT	†Field MRT	C_s	Lab MRT	†Field MRT	C_r	C_r /SOC
	Mg ha^{-1}	mg kg^{-1}	(%)	mg kg^{-1}	------ d ------		mg kg^{-1}	------ y ------		mg kg^{-1}	%
Switchgrass	0	252 (15) a	5.9	19	11	28	2161	3.5	8.7	2120 (100) a	49.3
	9.8	258 (6) a	5.6	38	21	52	1952	3.7	9.1	4473 (120) b	69.6
	19.5	270 (13) a	4.0	50	21	52	1755	3.0	7.5	5989 (80) c	79.3
	39.0	337 (15) b	3.1	86	23	57	1344	2.8	7.0	12303 (560) d	89.1
Digested	0	252 (15) a	5.9	19	11	28	2161	3.5	8.7	2120 (100) a	49.3
Fiber	9.8	270 (10) ab	5.8	40	19	47	1946	3.4	8.6	4897 (400) b	70.8
	19.5	286 (6) b	4.1	49	22	54	1749	2.9	7.1	7169 (250) c	80.5
	39.0	361 (19) c	3.0	72	20	51	1343	2.8	7.0	11328 (960) d	90.0
Softwood	0	252 (15) a	5.9	19	11	28	2161	3.5	8.7	2120 (100) a	49.3
Bark	9.8	251 (11) a	4.8	28	17	43	2114	4.1	10.2	3300 (290) b	69.7
	19.5	258 (6) a	3.3	33	22	55	2072	3.9	9.8	5130 (370) c	78.4
	39.0	292 (17) b	2.4	43	20	49	1987	3.2	8.0	9610 (200) d	86.6

Wood	0	252 (15) a	5.9	19	11	28	2161	3.5	8.7	2120 (100) a	49.3
Pellets	9.8	242 (6) a	3.8	18	20	49	2143	4.4	11.0	4390 (870) b	70.3
	19.5	248 (11) a	2.9	23	23	56	2120	4.3	10.7	6940 (830) c	78.9
	39.0	245 (14) a	1.8	27	23	58	2078	4.1	10.2	12870 (2080) d	86.8

†MRT-Mean residence times converted to field rates using a Q_{10} of 2; ($2^{(25-t)/10}$); where t is mean annual temperature = 11.9°C. §Values followed by the same letter within a column by treatment are not significantly different at P=0.05. Values between treatments followed by "*" within a depth increment are significantly different at P=0.05.

Table 22.D9. Pool sizes and C-mineralization kinetics of soil for the active and slow C pools for the Naff silt loam soil amended with 0, 9.8, 19.5 and 39 Mg ha^{-1} biochar.

		C-Mineralization		Active Pool			Slow Pool			Resistant Pool	
Cultivar	Rate	Cum. CO_2-C	C-min /SOC	C_a	Lab MRT	†Field MRT	C_s	Lab MRT	†Field MRT	C_r	C_r /SOC
	Mg ha^{-1}	mg kg^{-1}	(%)	mg kg^{-1}	------ d ------		mg kg^{-1}	------ y ------		mg kg^{-1}	%
Switchgrass	0	1645 (31) a	9.1	39	6	20	6424	1.7	5.2	11547 (600) a	64.1
	9.8	1690 (162) a	8.5	65	14	42	6208	1.8	5.6	16500 (1050) b	75.8
	19.5	1548 (84) a	6.8	118	19	59	5970	1.9	5.8	20820 (1580) c	82.5
	39.0	1407 (14) b	5.1	167	21	64	5546	1.7	5.3	22330 (1850) c	79.6
Digested	0	1645 (31) a	9.1	39	6	20	6424	1.7	5.2	11547 (600) a	64.1
Fiber	9.8	1658 (108) a	8.0	79	18	55	6190	1.6	4.9	14900 (1450) b	70.8
	19.5	1908 (21) b	8.3	54	9	28	6027	1.3	4.0	17500 (640) c	75.0
	39.0	2047 (64) c	7.7	47	7	21	5651	1.1	3.5	23190 (3960) d	83.4

Softwood	0	1645 (31) a	9.1	39	6	20	6424	1.7	5.2	11547 (600) a	64.1
Bark	9.8	1717 (63) ab	8.0	52	15	46	6373	1.8	5.5	15550 (940) b	71.0
	19.5	1800 (19) b	7.8	32	9	28	6356	1.5	4.6	18220 (650) c	75.9
	39.0	1927 (41) c	6.4	18	4	12	6295	1.3	4.0	26430 (2050) d	82.9
Wood	0	1645 (31) a	9.1	39	6	20	6424	1.7	5.2	11547 (600) a	64.1
Pellets	9.8	1700 (70) a	8.2	32	10	31	6412	1.6	5.0	17850 (1510) b	77.4
	19.5	1678 (73) a	6.9	52	17	54	6374	1.6	5.1	23730 (130) c	85.4
	39.0	1742 (108) a	5.1	95	33	102	6293	1.6	5.1	30040 (3090) d	83.4

†MRT-Mean residence times converted to field rates using a Q_{10} of 2; ($2^{(25-t)/10}$); where t is mean annual temperature = 8.7 °C. §Values followed by the same letter within a column by treatment are not significantly different at P=0.05. Values between treatments followed by "*" within a depth increment are significantly different at P=0.05.

Table 22.D10. Pool sizes and C-mineralization kinetics of soil for the active and slow C pools for the Palouse silt loam soil amended with 0, 9.8, 19.5 and 39 Mg ha^{-1} biochar.

		C-Mineralization		Active Pool			Slow Pool			Resistant Pool	
Cultivar	Rate	Cum. CO_2-C	C-min /SOC	C_a	Lab MRT	†Field MRT	C_s	Lab MRT	†Field MRT	C_r	C_r /SOC
	Mg ha^{-1}	mg kg^{-1}	(%)	mg kg^{-1}	------ d ------		mg kg^{-1}	------ y ------		mg kg^{-1}	%
Switchgrass	0	1958 (32) ab	8.5	203	26	81	6111	1.2	3.6	16846 (100) a	72.7
	9.8	1895 (55) a	7.3	159	19	58	5965	1.2	3.8	22370 (5850) a	81.0
	19.5	2017 (70) ab	7.1	152	16	51	5787	1.0	3.2	23900 (4080) a	81.5
	39.0	2051 (67) b	6.4	178	14	44	5386	1.0	3.0	27770 (5380) a	84.7
Digested	0	1958 (32) a	8.5	203	26	81	6111	1.2	3.6	16846 (100) a	72.7
Fiber	9.8	2057 (47) b	8.1	176	19	60	5944	1.0	3.1	19600 (1940) ab	76.5
	19.5	2020 (253) ab	6.9	160	18	57	5772	1.0	3.2	23500 (2270) b	79.4
	39.0	2314 (49) c	6.1	140	11	35	5409	0.8	2.3	29960 (1620) c	80.8
Softwood	0	1958 (32) a	8.5	203	26	81	6111	1.2	3.6	16846 (100) a	72.7
Bark	9.8	1957 (51) a	6.5	200	23	70	6076	1.1	3.5	21880 (2150) b	74.0
	19.5	2143 (45) b	6.0	182	23	72	6057	1.0	3.1	23290 (1380) b	71.2
	39.0	2102 (97) ab	5.8	196	27	84	5968	1.0	3.1	29360 (2290) c	81.9

Wood	0	1958 (32) ab	8.5	203	26	81	6111	1.2	3.6	16846 (100) a	72.7
Pellets	9.8	2020 (30) bc	7.7	201	23	71	6094	1.1	3.5	21090 (1060) b	78.0
	19.5	1940 (32) a	5.6	195	25	77	6082	1.2	3.7	28840 (2000) c	80.6
	39.0	2062 (57) c	5.3	172	21	65	6067	1.1	3.6	30820 (2120) c	81.3

†MRT-Mean residence times converted to field rates using a Q_{10} of 2; ($2^{(25-t)/10}$); where t is mean annual temperature = 8.7 °C. [§]Values followed by the same letter within a column by treatment are not significantly different at P=0.05. Values between treatments followed by "*" within a depth increment are significantly different at P=0.05.

Table 22.D11. Pool sizes and C-mineralization kinetics of soil for the active and slow C pools for the Thatuna silt loam soil amended with 0, 9.8, 19.5 and 39 Mg ha^{-1} biochar.

		C-Mineralization		Active Pool			Slow Pool			Resistant Pool	
Cultivar	Rate	Cum. CO_2-C	C-min /SOC	C_a	Lab MRT	†Field MRT	C_s	Lab MRT	†Field MRT	C_r	C_r /SOC
	Mg ha^{-1}	mg kg^{-1}	(%)	mg kg^{-1}	------ d ------		mg kg^{-1}	------ y ------		mg kg^{-1}	%
Switchgrass	0	2264 (57) a	8.4	388	30	93	10049	1.9	5.8	16423 (210) a	61.1
	9.8	2482 (67) b	8.8	333	31	96	9914	1.7	5.4	21830 (2060) b	70.8
	19.5	2317 (67) a	7.5	370	50	155	9692	1.8	5.6	26300 (2020) c	76.3
	39.0	2332 (32) a	6.2	314	37	116	9373	1.9	5.7	29460 (2340) c	75.6
Digested	0	2264 (57) a	8.4	388	30	93	10049	1.9	5.8	16423 (210) a	61.1
Fiber	9.8	2749 (37) b	9.3	293	27	83	9950	1.5	4.7	19390 (830) b	75.6
	19.5	2682 (65) b	8.8	296	24	73	9759	1.5	4.7	26970 (1700) c	78.4
	39.0	3013 (70) c	8.1	253	19	59	9419	1.3	4.0	28200 (230) c	74.8
Softwood	0	2264 (57) a	8.4	388	30	93	10049	1.9	5.8	16423 (210) a	61.1
Bark	9.8	2156 (120) a	7.6	386	29	90	10013	1.9	5.7	19460 (790) b	66.7
	19.5	2590 (71) b	8.1	266	25	78	10096	1.6	5.0	19950 (2080) b	65.3
	39.0	2787 (85) c	8.0	297	28	88	9990	1.4	4.5	26340 (1780) c	74.1

Wood	0	2264 (57) a	8.4	388	30	93	10049	1.9	5.8	16423 (210) a	61.1
Pellets	9.8	2064 (81) b	7.4	402	61	189	10016	2.2	6.8	18480 (1390) a	65.1
	19.5	2214 (32) a	7.3	294	4	11	10106	2.0	6.2	23460 (1550) b	72.7
	39.0	2054 (206) ab	5.7	423	71	220	9939	2.3	7.3	33350 (1990) c	79.7

†MRT-Mean residence times converted to field rates using a Q_{10} of 2; ($2^{(25-t)/10}$); where t is mean annual temperature = 8.7 °C. §Values followed by the same letter within a column by treatment are not significantly different at P=0.05. Values between treatments followed by "*" within a depth increment are significantly different at P=0.05.

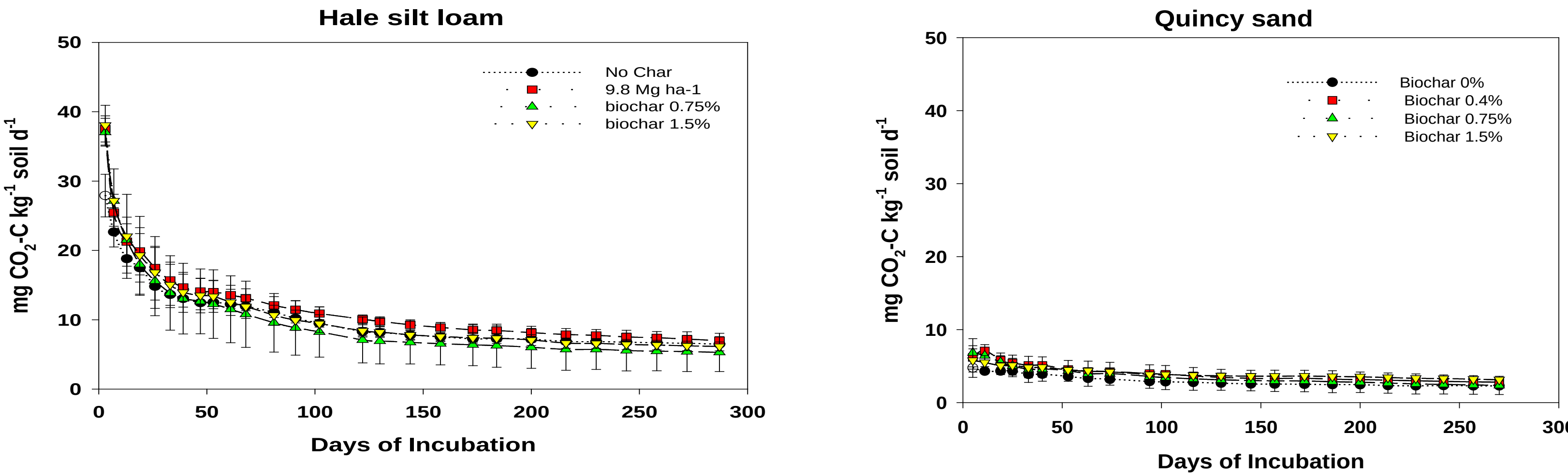
Table 22.D12. Pool sizes and C-mineralization kinetics of soil for the active and slow C pools for the Hale silt loam soil amended with 0, 9.8, 19.5 and 39 Mg ha^{-1} biochar.

		C-Mineralization		Active Pool			Slow Pool			Resistant Pool	
Cultivar	Rate	Cum. CO_2-C	C-min /SOC	C_a	Lab MRT	†Field MRT	C_s	Lab MRT	†Field MRT	C_r	C_r /SOC
	Mg ha^{-1}	mg kg^{-1}	(%)	mg kg^{-1}	------ d ------		mg kg^{-1}	------ y ------		mg kg^{-1}	%
Switchgrass	0	1844 (64) a	4.6	418	19	59	10282	2.3	7.1	29180 (990) a	73.2
	9.8	1866 (79) a	4.2	430	17	54	10080	2.2	6.8	30360 (110) a	72.2
	19.5	1844 (66) a	4.2	399	16	50	9926	2.2	6.7	34210 (1160) b	77.1
	39.0	1686 (45) b	3.5	449	15	47	9501	2.3	7.1	41050 (1770) c	82.0
Digested	0	1844 (64) a	4.6	418	19	59	10282	2.3	7.1	29180 (990) a	73.2
Fiber	9.8	1830 (81) a	4.3	399	16	49	10107	2.2	6.9	31170 (770) a	74.2
	19.5	1743 (66) a	3.9	462	20	62	9856	2.3	7.0	34190 (1340) b	76.8
	39.0	1863 (106) a	3.7	550	17	54	9385	2.0	6.1	41620 (1200) c	81.7
Softwood	0	1844 (64) a	4.6	418	19	59	10282	2.3	7.1	29180 (990) a	73.2
Bark	9.8	1873 (94) a	4.4	467	22	67	10195	2.3	7.0	31040 (1060) ab	74.1
	19.5	1795 (57) a	4.0	397	15	47	10228	2.3	7.3	32110 (670) b	74.4
	39.0	1845 (53) a	3.7	363	13	41	10187	2.3	7.0	41940 (1050) c	81.8

Wood	0	1844 (64) a	4.6	418	19	59	10282	2.3	7.1	29180 (990) a	73.2
Pellets	9.8	1854 (70) a	4.3	486	22	68	10195	2.2	7.0	34120 (3050) ab	76.9
	19.5	1774 (30) a	3.9	343	12	37	10320	2.4	7.4	35520 (2620) b	77.2
	39.0	1783 (81) a	3.5	314	12	38	10311	2.4	7.6	44870 (1360) c	83.3

†MRT-Mean residence times converted to field rates using a Q_{10} of 2; ($2^{(25-t)/10}$); where t is mean annual temperature = 9.6 ^{o}C. §Values followed by the same letter within a column by treatment are not significantly different at P=0.05. Values between treatments followed by "*" within a depth increment are significantly different at P=0.05.

CARBON MINERALIZATION- PEANUT HULL BIOCHAR



ACTIVATED CHARCOAL

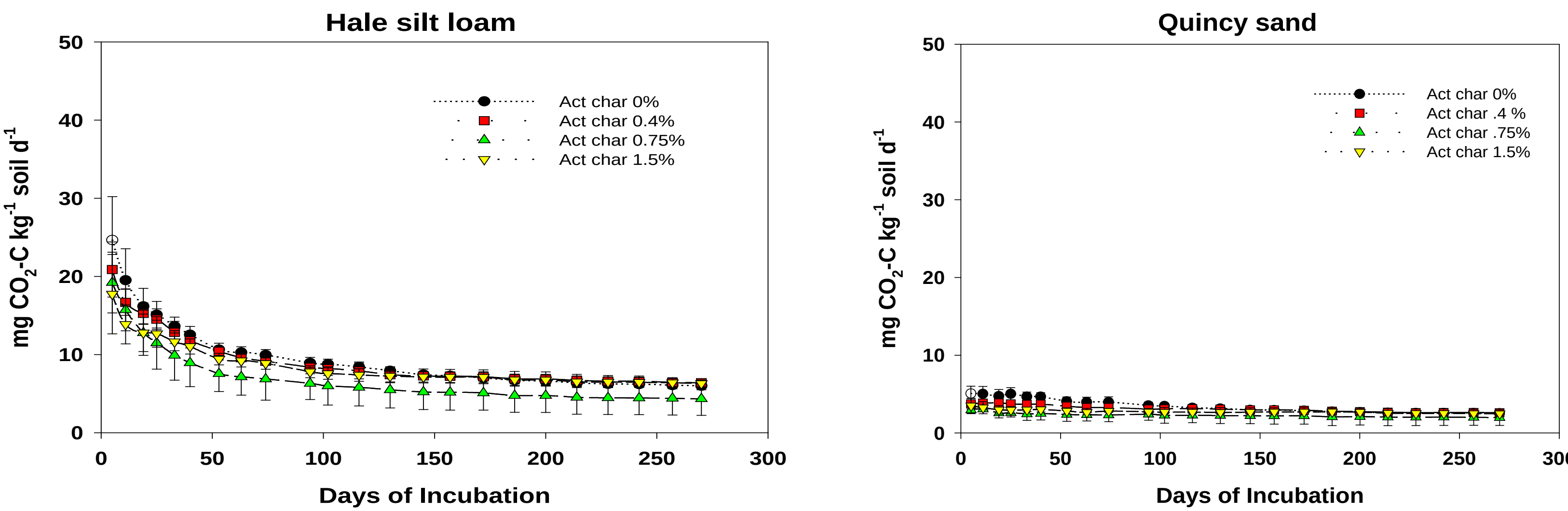


Figure 22.D1. Soil C-mineralization rates for the Quincy sand and Hale silt loam soils incubated with peanut hull biocharand activated carbon amendment.

Quincy Sand

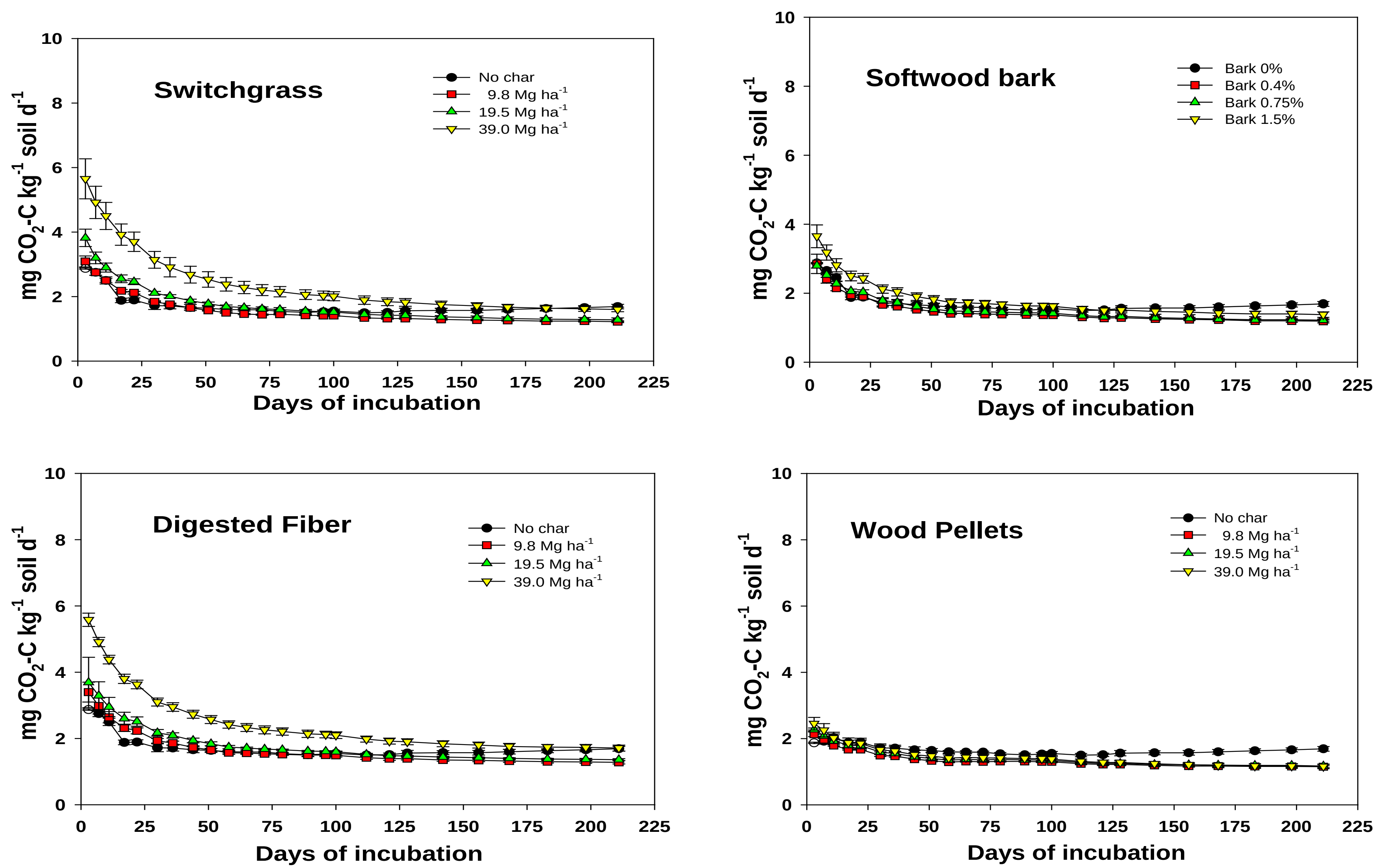


Figure 22.D2. Soil C-mineralization rates for the Quincy Sand incubated with biochar amendments. The biochars were made at a pyrolysis temperature of 500°C.

Naff Silt loam

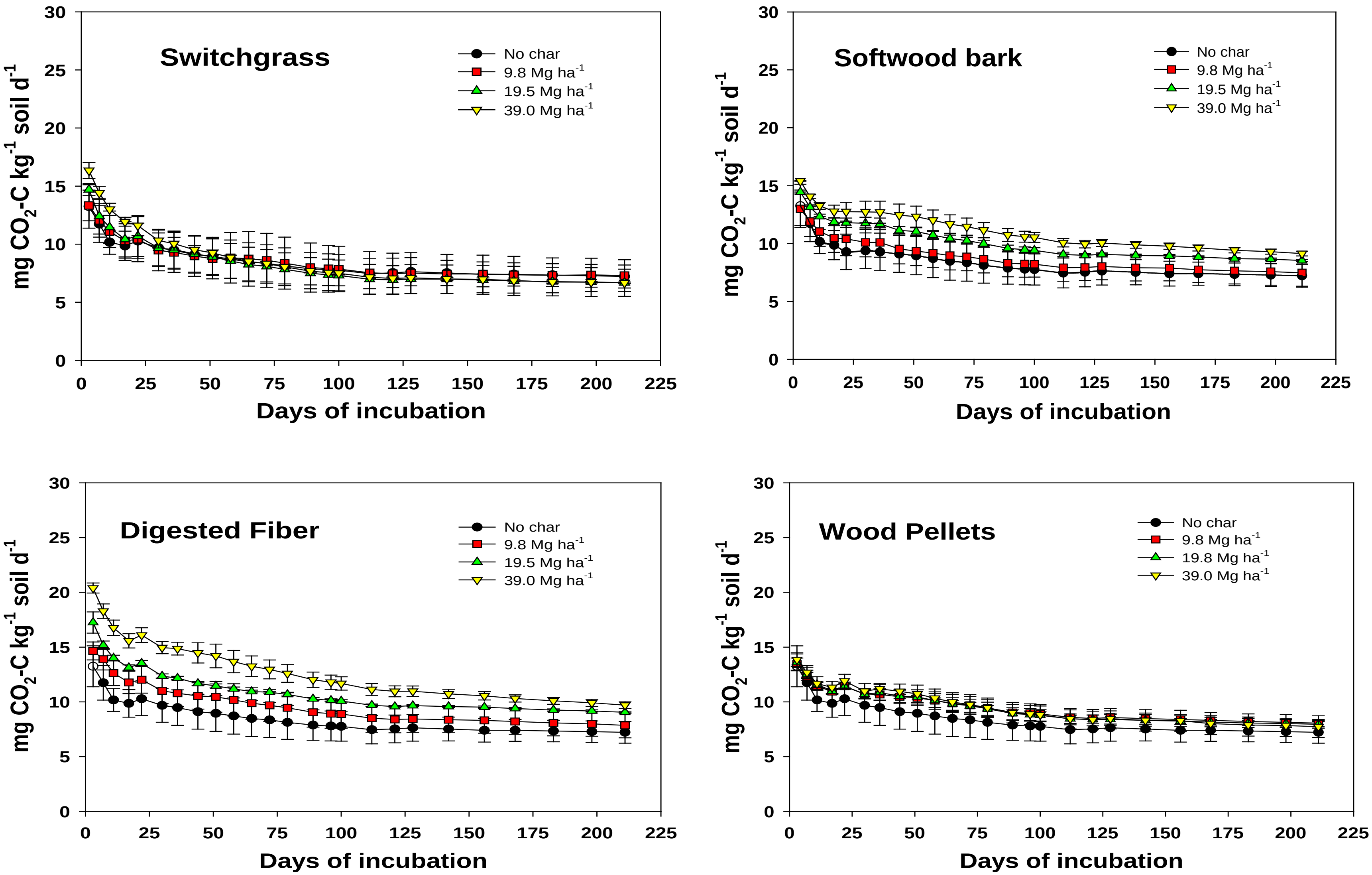


Figure 22.D3. Soil C-mineralization rates for the Naff Silt loam incubated with biochar amendments. The biochars were made at a pyrolysis temperature of 500°C.

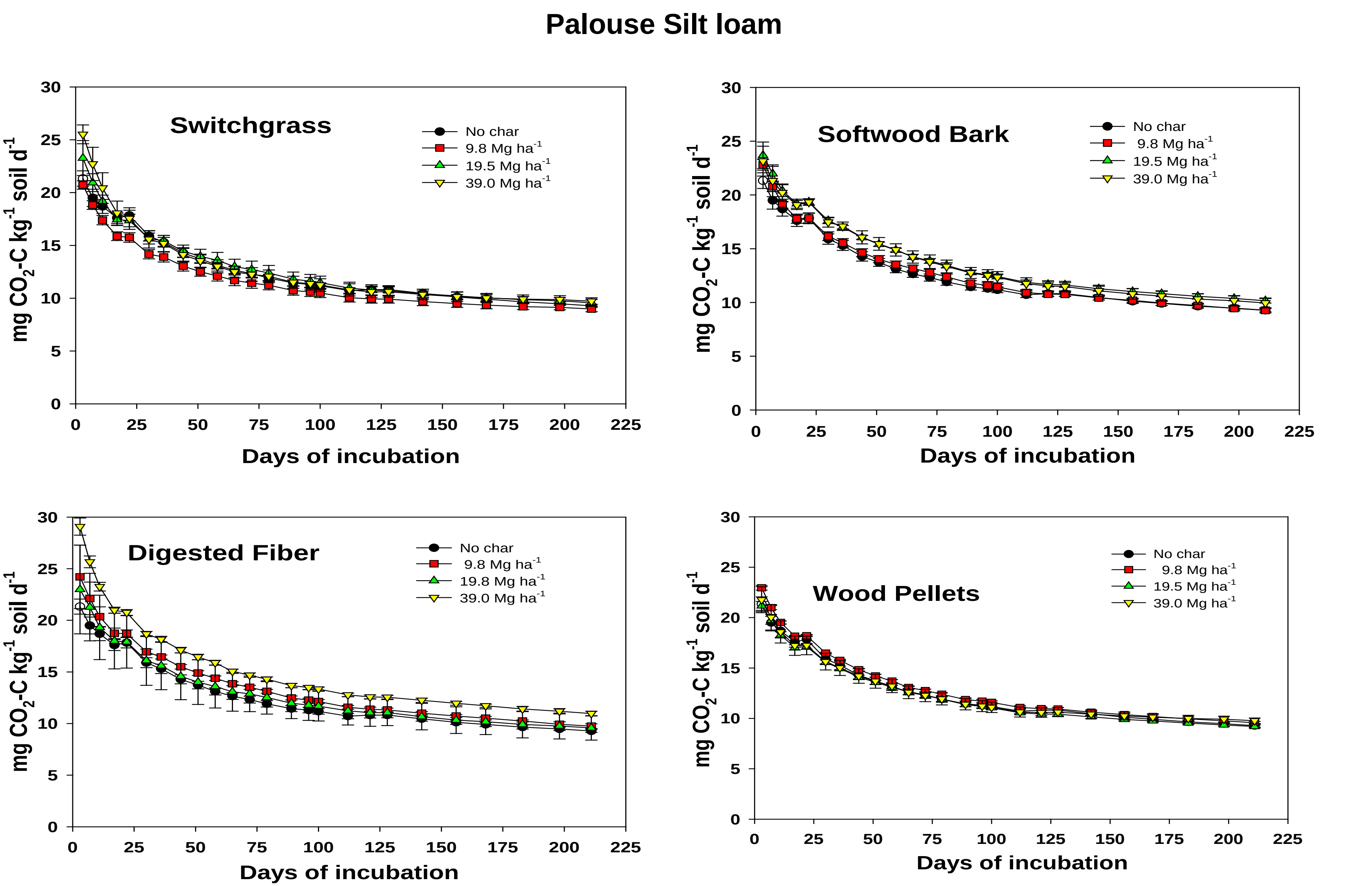


Figure 22.D4. Soil C-mineralization rates for the Palouse Silt loam incubated with biochar amendments. The biochars were made at a pyrolysis temperature of 500°C.

Thatuna Silt loam

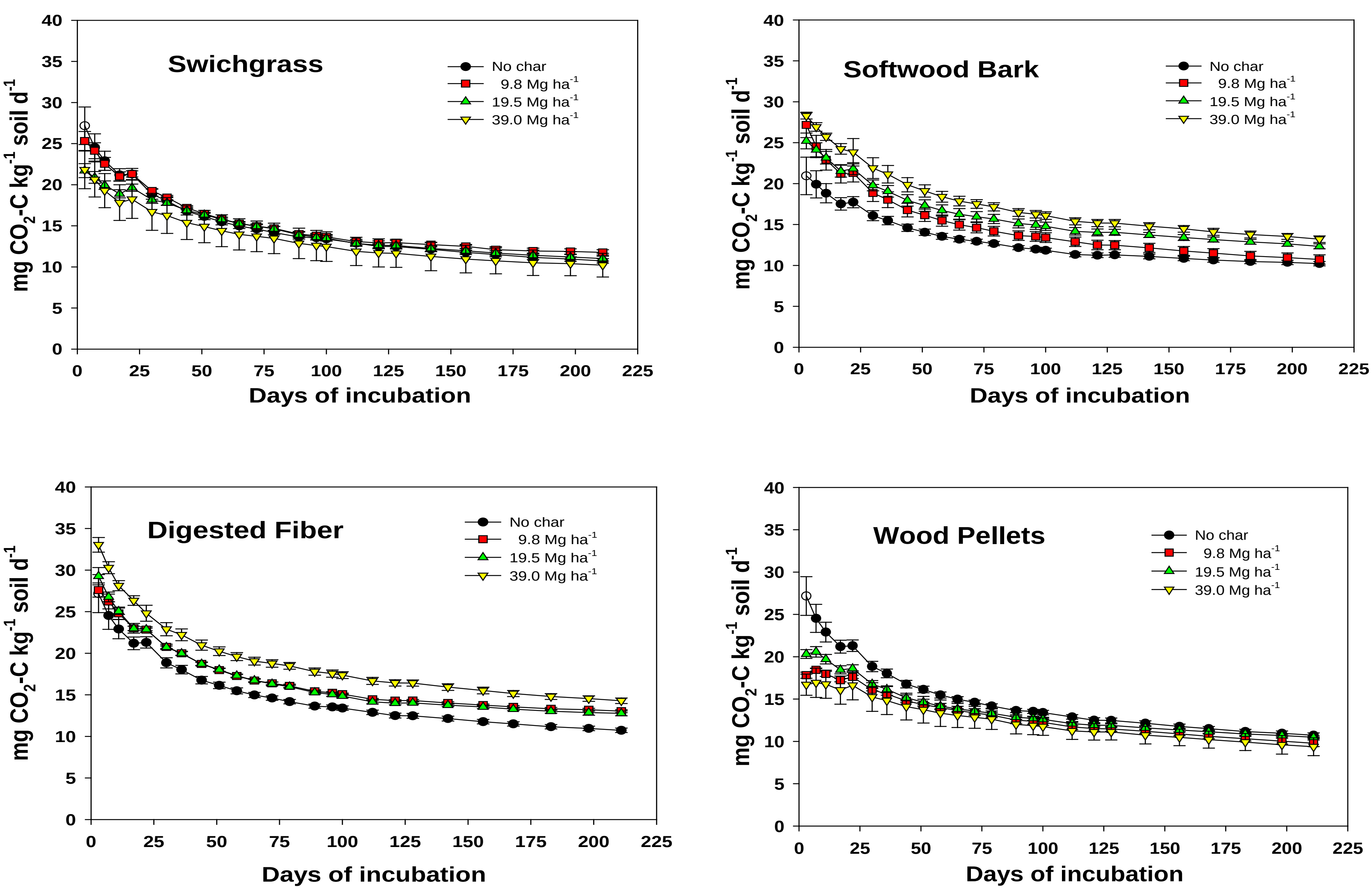


Figure 22.D5. Soil C-mineralization rates for the Thatuna Silt loam incubated with biochar amendments. The biochars were made at a pyrolysis temperature of 500ºC.

Hale Silt loam

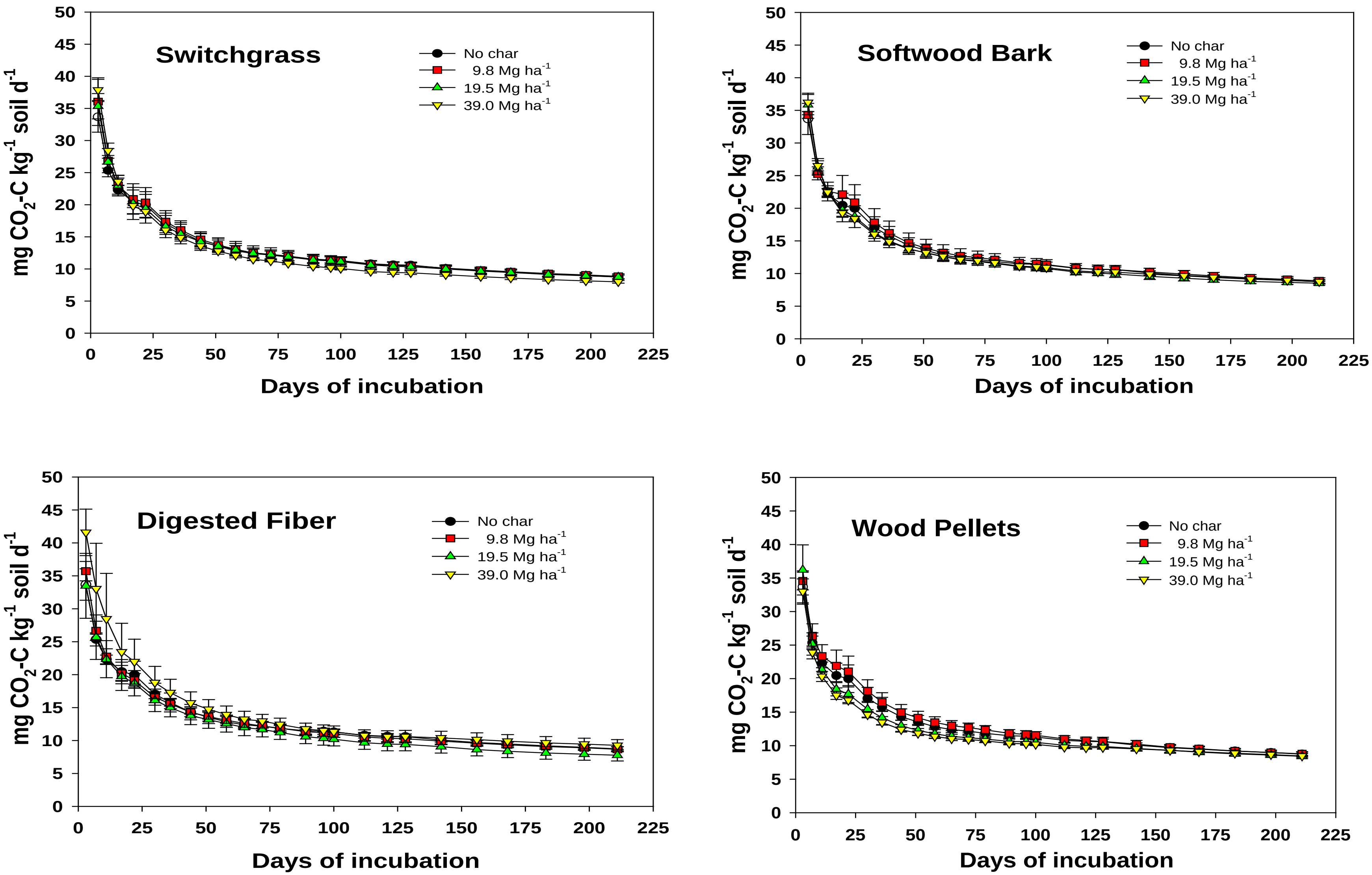


Figure 22.D6. Soil C-mineralization rates for the Hale Silt loam incubated with biochar amendments. The biochars were made at a pyrolysis temperature of 500ºC.

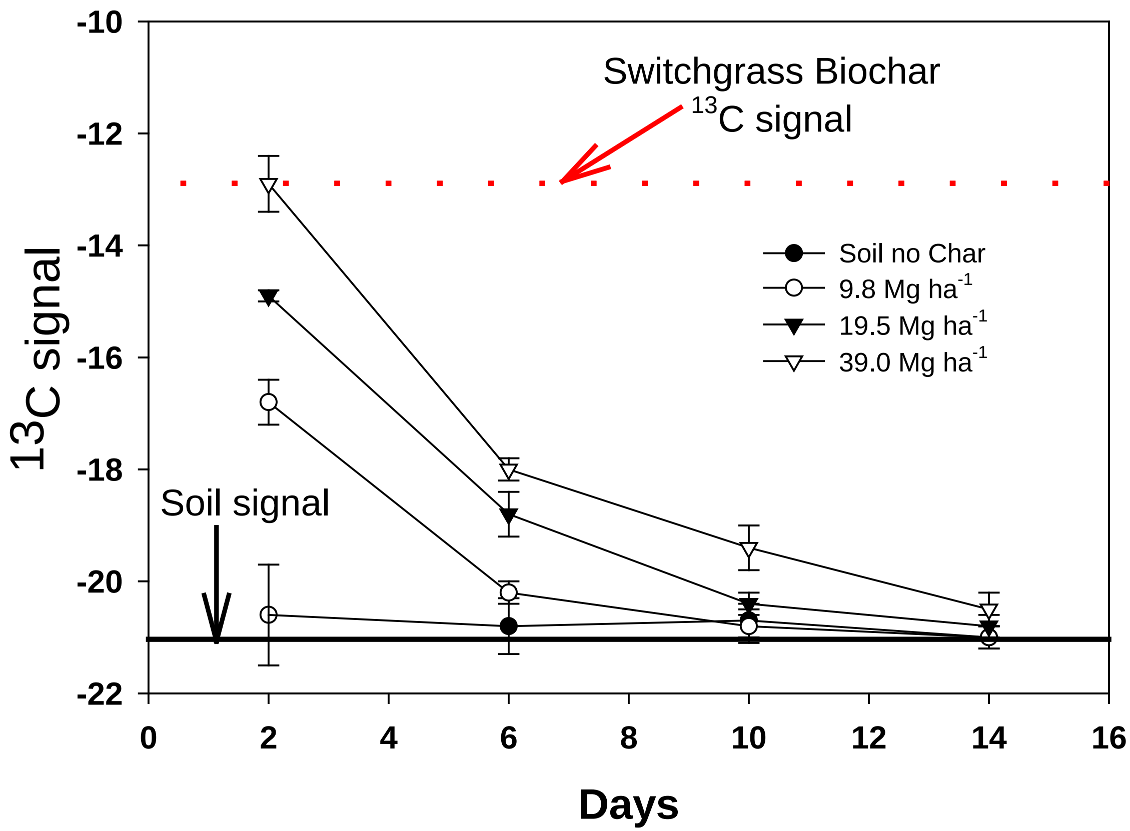


Figure 22.D7. δ ^{13}C signals from soil, switchgrass biochar and soil biochar mixes. The dashed line represents the δ ^{13}C signal from the biochar and the solid line the background δ ^{13}C of the soil. The symbols represent the different rates of biochar additions. The signal within 14 days returns to the soil background indicating the majority of the initial flush of CO_2 originated from the biochar and not the native soil organic matter.

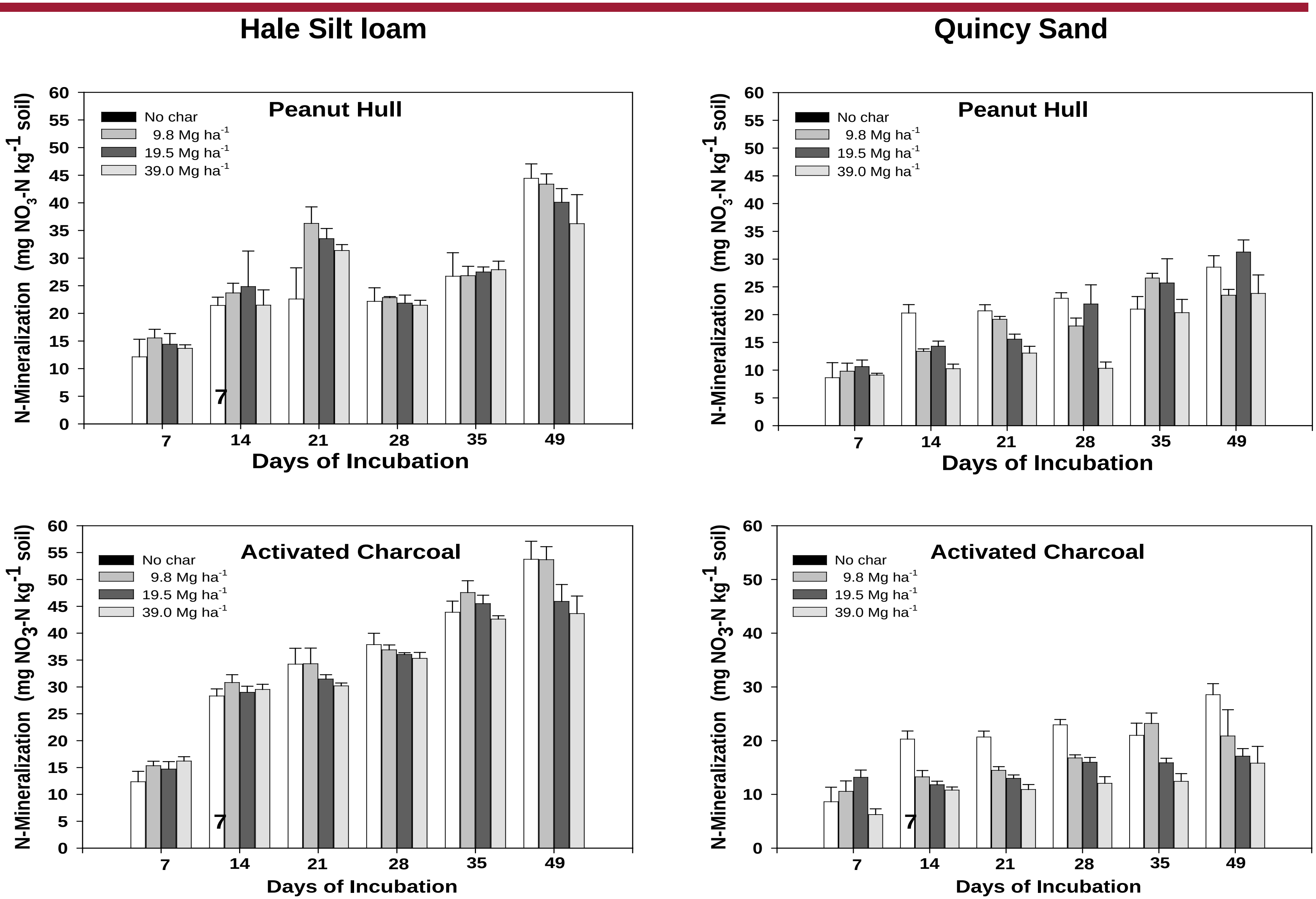


Figure 22.D8. Soil N-mineralization rates for the Hale silt loam and Quincy sand incubated with peanut hull biochar and activated charcoal amendments. The biochar was made at the pyrolysis temperature of 500°C.

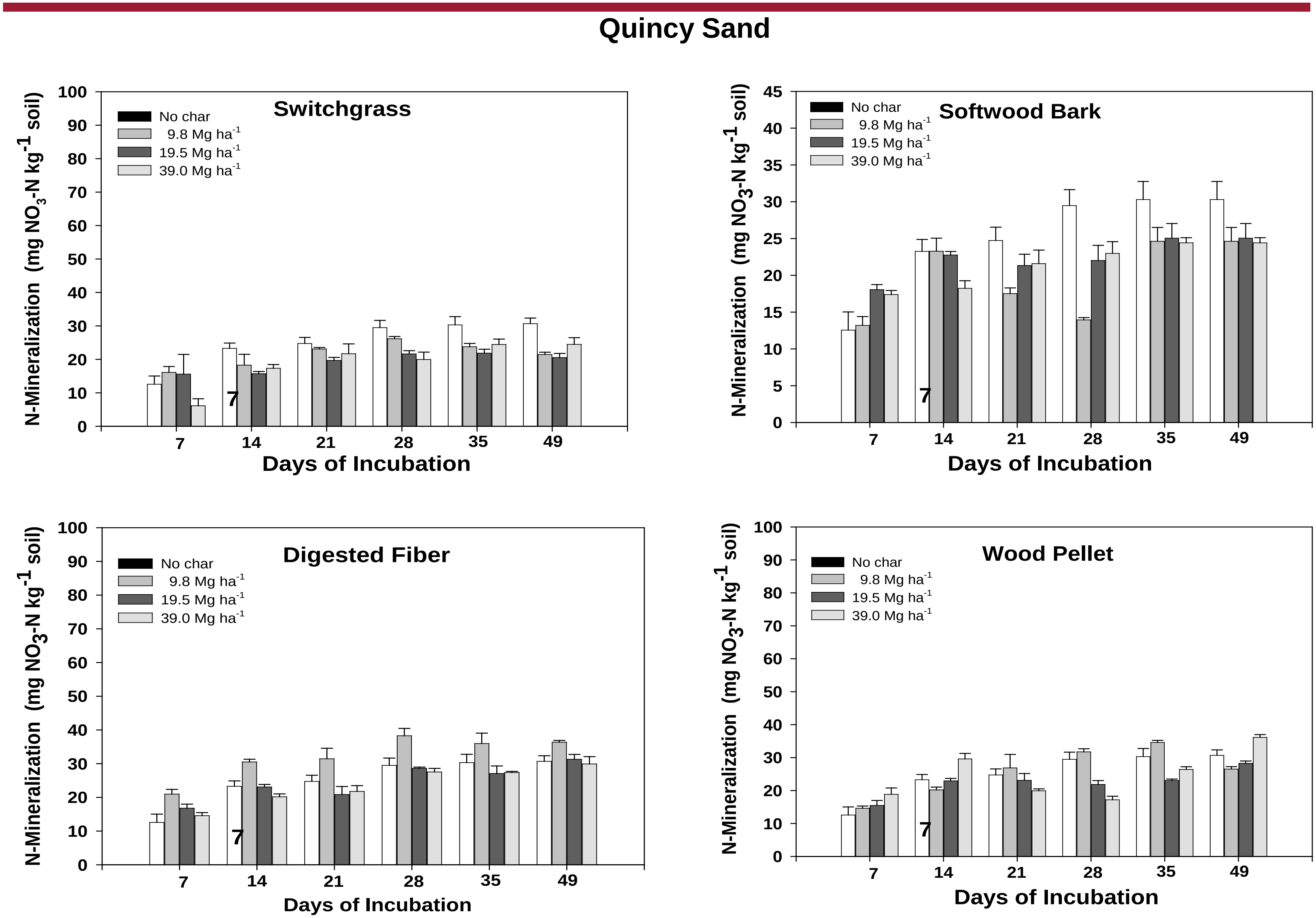


Figure 22.D9. Soil N-mineralization rates for the Quincy sand incubated with biochar amendments. The biochars were made at the pyrolysis temperature of 500°C.

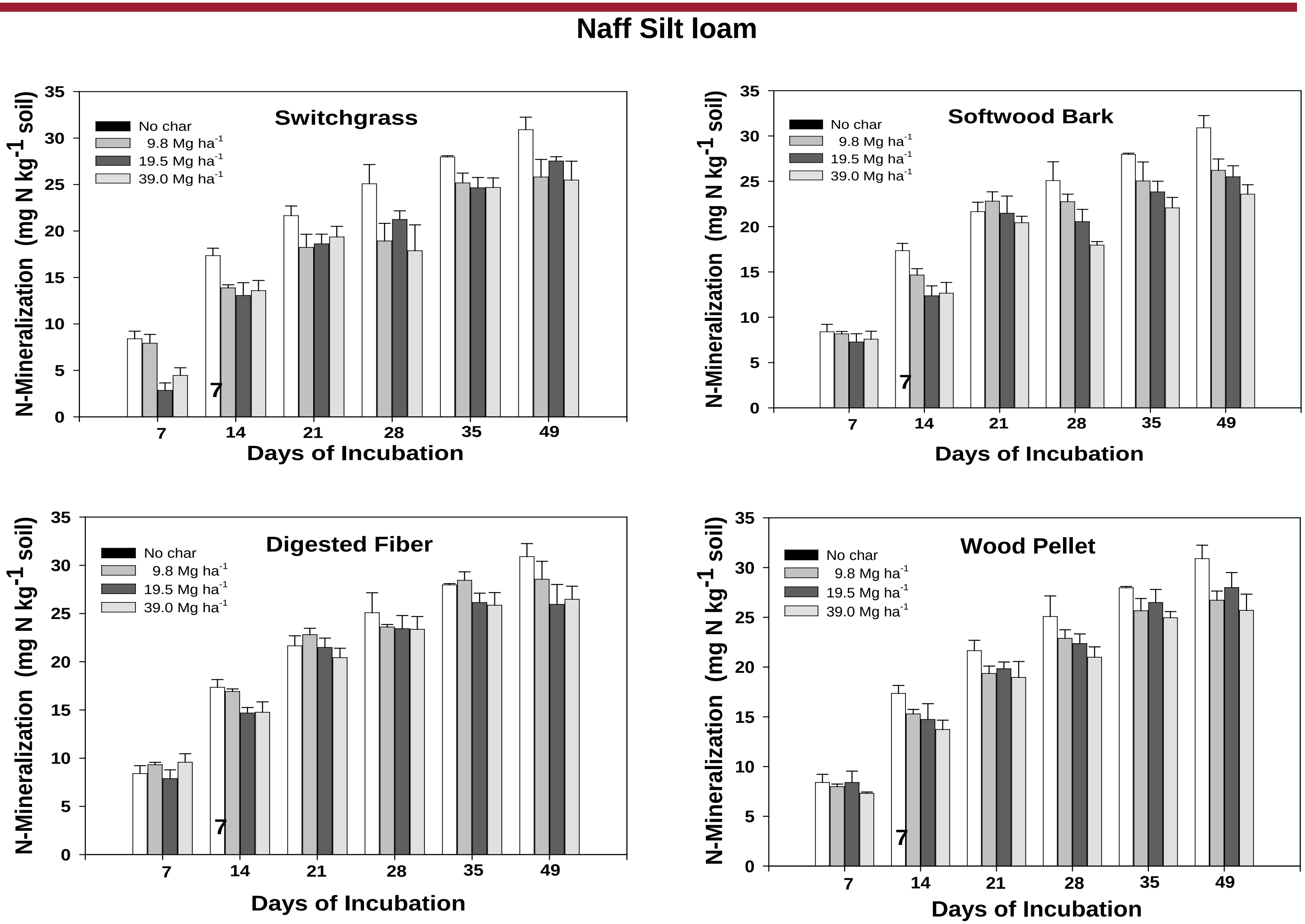


Figure 22.D10. Soil N-mineralization rates for the Naff silt loam incubated with biochar amendments. The biochars were made at the pyrolysis temperature of 500ºC.

Palouse Silt loam

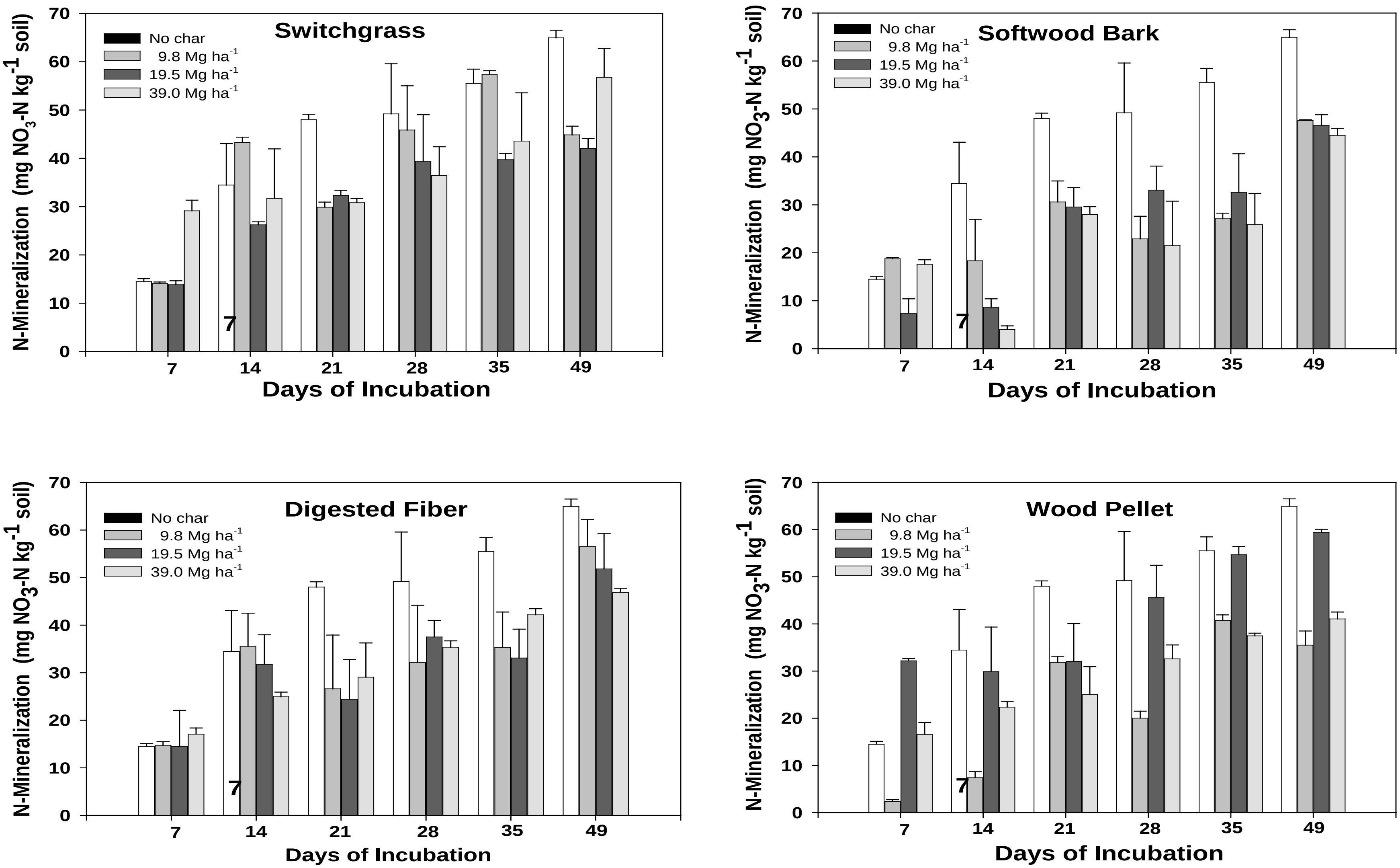


Figure 22.D11. Soil N-mineralization rates for the Palouse silt loam incubated with biochar amendments. The biochars were made at the pyrolysis temperature of 500°C.

Thatuna Silt loam

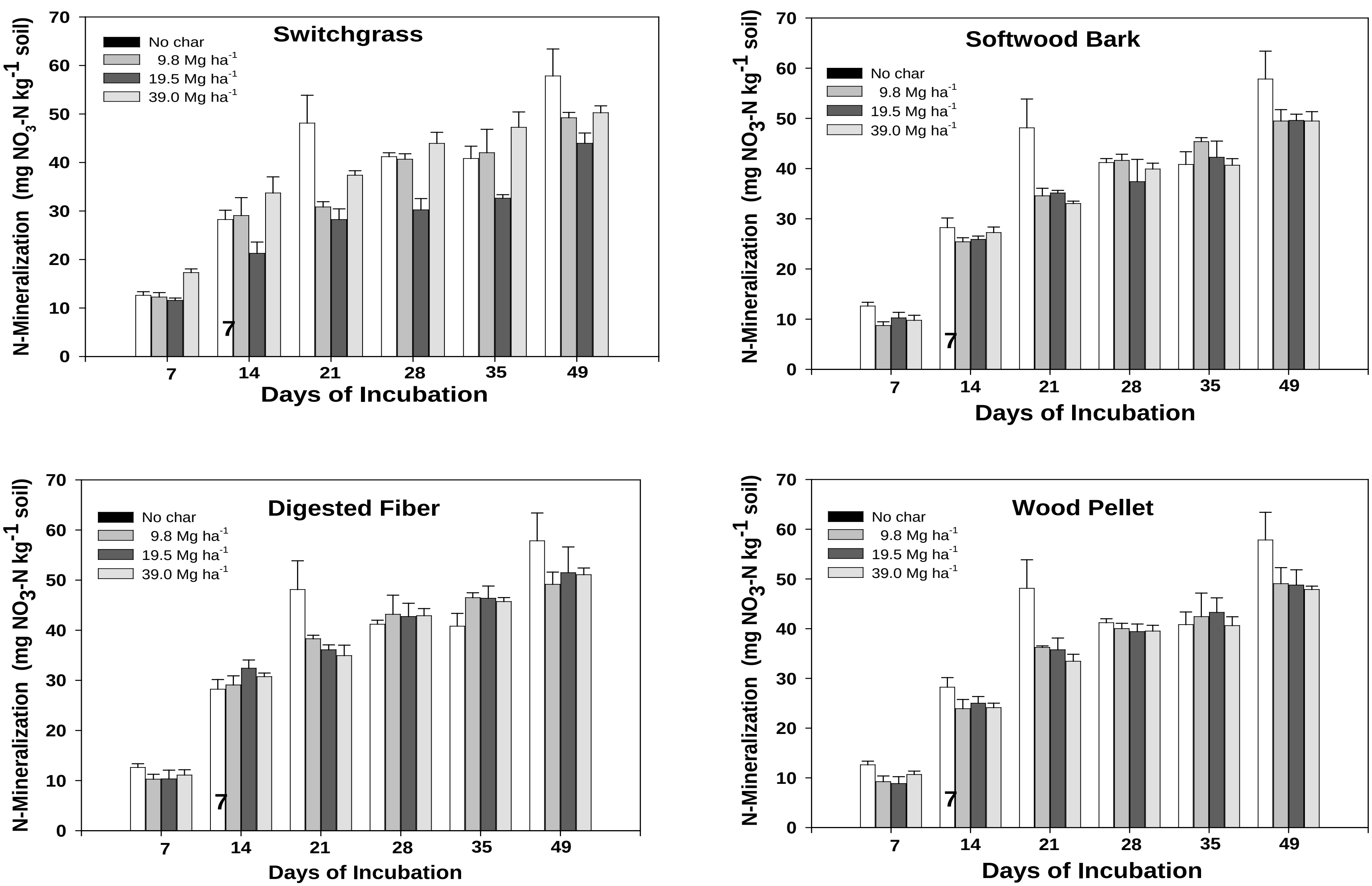


Figure 22.D12. Soil N-mineralization rates for the Thatuna silt loam incubated with biochar amendments. The biochars were made at the pyrolysis temperature of 500°C.

Hale Silt Loam

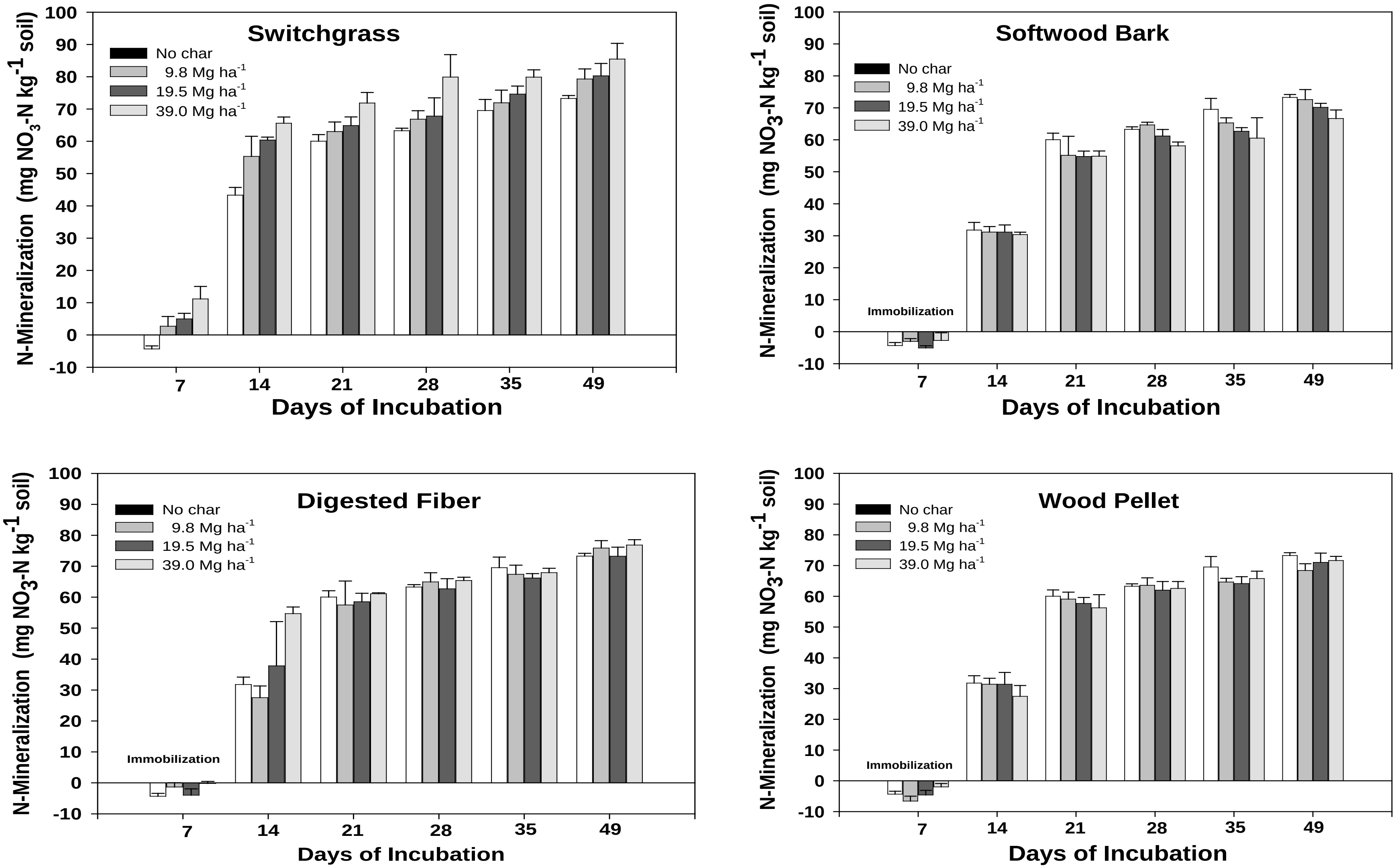


Figure 22.D13. Soil N-mineralization rates for the Hale silt loam incubated with biochar amendments. The biochars were made at a pyrolysis temperature of 500°C.