Swan View Coalition *Nature and Human Nature on the Same Path*

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September 30, 2023

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PDF submitted via https://cara.fs2c.usda.gov/Public/CommentInput?project=63176
and to gary.blazejewski@usda.gov

Re: Dry Riverside EA

Dear Mr. Blazejewski;

Please accept these comments in the above matter into the public record. We incorporate by reference the comments being submitted by Friends of the Wild Swan and those submitted by WildEarth Guardians in this matter. We do not see where the EA has adequately addressed any of the issues we raised during Scoping, so we incorporate our 12/19/22 Scoping comments into this letter as well. Due to the size and scope of this Project and the substantial uncertainty regarding the effects of the PA, we feel the effects must be analyzed and disclosed in an Environmental Impact Statement.

Road Building and Closure Effectiveness

The EA relies on an outdated description of its road closure effectiveness monitoring (PF Exhibit Q-17, Kuennen 2019). It then relies on outdated effectiveness data (2019-2020) to conclude Forest-wide effectiveness of 92% (EA page 54), as does the revised Biological Opinion on the Flathead Forest Plan (Plan BiOp). It then provides absolutely no inspection data to support its conclusion that “there are no known persistent ineffective closures within the analysis area,” nor any quantitative or other definition of “persistent.” It then provides the same boilerplate language used in the Plan BiOp to discount, rather than account for, the effects of ineffective road closures on grizzly bears (EA at 54-56).

This boilerplate discounting of the effects on grizzly bears has been deemed inadequate and unlawful in a number of U.S. District Court rulings on the Kootenai NF and on the Helena-Lewis and Clark NF. See, for example, the Kootenai NF Knotty Pine decision at
http://www.swanview.org/reports/Knotty-Pine-prelininary-injunction-order-04242023.pdf
and the Helena-Lewis and Clark NF decision at:
http://www.swanview.org/reports/Helena_illegal_roads_order_filed_8.03.23.pdf .

It does no good for the EA to add (page 54) that "Effects of past illegal use of roads on grizzly bears are part of the baseline conditions that have supported the expanding population and distribution of grizzly bears in the NDCE recovery zone" and deem the problem "inconsequential." The Flathead is adding more miles to its road System as "impassable" by not counting them in calculations of TMRD, even though they will continue to function as roads – thus increasing the number of roads and the number of ineffective road closures over what was included in the 2011 baseline. See the above court orders and our discussion that follows.

Please also see our 2023 "Road Hunt" road closure effectiveness report based on data collected while inspecting 303 FS road closures in the Swan Valley Geographic Area in 2022. (https://www.swanview.org/reports/Road_Hunt_Hammer_2023.pdf and submitted separately via email to Gary Blazejewski). We found only 53% of the closures showed no sign of motorized vehicles behind them and, after allowing for administrative and logging contractor use, found that effectiveness rose to only 68%. Our report discusses reasons for the disparity between the Flathead's previous finding of 92% effectiveness, shows that the Flathead's 2021 and 2022 data indicate a lower effectiveness, discusses flaws in the Flathead's road closure monitoring program, demonstrates how road closure violations can persist for many years before the closure device is repaired, reports on how dense vegetation contributes to road closure effectiveness, and discusses how the Flathead has not followed through with promises made to FWS during consultation on the revised Plan BiOp.

The EA and Project File documents located on the Project web site rely on old data and procedures and do not use the best available data and science available, as required by law. Some of that best available data would be the Flathead's own 2021 – 2023 road closure effectiveness data as well as our 2022 Swan Valley GA data and 2023 report. Moreover, the EA must include detailed road closure effectiveness data for each road in the analysis area, including when each closure has been inspected and whether it was found effective or not, and a clear accounting of how rebuilding previously abandoned or decommissioned roads (often overgrown with vegetation) and simply closing them as "impassable" lowers closure effectiveness and grizzly bear security.

The Project would rebuild 21.5 miles of previously abandoned or decommissioned roads and return them to the road System as "impassable to motorized vehicles" after blocking as little as the first 50′ of the road – plus 1.2 miles of newly constructed road and 5.2 miles of "temporary" roads. Our Road Hunt report provides visual examples of where motor vehicles are detouring around closure devices for distances in excess of 50′ (page 17). The EA needs to detail the current condition of each of the non-system roads intended to be rebuilt, including its ability to resist motorized trespass, then detail to what degree rebuilding each road and simply rendering it "impassable" will increase its vulnerability to motorized trespass.

We find the revised BiOp to suffer the same legal inadequacies Judge Molloy found in the 2017 BiOp, especially in regards to the abandonment of Amendment 19's requirements. The revised BiOp emphasizes several times in bold face that the Forest Plan and its implementing projects will and must maintain the 2011 **"on the ground"** grizzly bear habitat conditions. Yet it still allows the construction of new roads and the reconstruction of old roads without them showing up/being counted in TMRD.

Projects like Dry Riverside, for example, can build new roads and rebuild historic roads (even through Secure Core) and then simply close them as "impassable" roads - rather than have to reclaim and/or decommission them in order to omit them from TMRD. Rebuilding historic road templates in this Project and then simply rendering the road "impassable" to motor vehicles for the first 50' does not provide the grizzly bear security that the previous status of historic road and "existing template" provided. Brand new road templates and old templates newly cleared of vegetation do not provide the previously existing impediments to human travel nor the resulting **"on the ground"** habitat conditions and security that previously existed for grizzly bear.

By not requiring that "impassable" roads be included in TMRD, the Project, Plan and Plan BiOp allow unlimited miles of roads to be constructed without increasing TMRD above 2011 levels. While this sleight of hand may maintain 2011 numbers, it most certainly does not maintain 2011 "**on the ground** habitat conditions" and habitat security – premises and promises upon which the Plan and its BiOps are based.

This sleight of hand is perpetuated in the EA, where it notes on page 57:

> To meet FW-STD-IFS-02, upon completion of project activities, all temporary roads would be rehabilitated following timber harvest activities and would cease to function as roads. New roads (22.7 miles) would be added to the National Forest Service Road System (NFS) and treated to be impassable (as defined in the forest plan) to wheeled motorized vehicles.

Hidden in this statement is the presumption that, by not increasing public wheel-driven motorized access, the project will not increase public non-motorized access – which is simply not true. Indeed, the EA (page 59) warns that "Disturbance from motorized access, mechanical **and human activity** would disturb grizzly bears potentially altering travel and foraging patterns. Motorized use **and human activity** have the potential to cause grizzly bear mortality." (Emphasis added).

"Impassable" roads continue to function as roads for non-motorized public access that has documented negative impacts on grizzly bears. These roads also provide for additional impacts by wheel-driven motorized trespass of the "impassable" barrier and the lawful use of motorized over-snow vehicles. Page 60 of the EA indeed finds "Overgrown roads cleared for project activities may indirectly allow for easier winter snowmobile access in the project area." But those impacts are not accounted for by showing the actual increase in total road density/TMRD – they are instead dismissed/omitted as though the new roads don't exist and have no impacts.

Timing and Project Duration Impacts on Grizzly Bears

The EA (page 60) acknowledges that "Prescribed burn units within the Betty Baptiste Project may not be complete when implementation of Dry Riverside starts. Other activities such as helicopter use or implementation activities using motorized administrative access may overlap in the Dry Riverside subunit."

Project File Exhibit Q-006 attempts to show compliance with Plan Standard FW-STD-IFS-03, which limits temporary project increases in OMRD to 5%, increases in TMRD to 3%, and decreases in CORE to 2% - using a 10-year running average. The Exhibit, however, falls flat and does not adequately calculate or describe how it calculates these temporary increase percentages.

The footnote to Table 8 explains that, "For example, for Riverside Paint – OMRD – Yr 1-10, this is the average of Riverside Paint OMRD Year 1-10 in Table 7 above." And so it is, with a sum of 33 percentage points in Table 7 divided by 10 to arrive at a 3.3% 10-year average in Table 8. This calculation method, however, does not hold true for Year 1-10 for Riverside Paint CORE (0/10 = 0, not 1.0), or Logan Dry Park OMRD (40/10 = 4.0, not 3.8), TMRD (11/10 = 0.11, not 0.10), or CORE (-15/10 = -1.5, not +1.5). From there it just gets worse as one moves forward a year at a time and is left wondering if decreases in CORE are expressed as positive or negative numbers and whether the values in Table 7 column Year 12 carry on out through "yr 12-21" in Table 8. This is not explained anywhere, nor is it explained why 2019 was chosen as the starting point for this exercise.

Forest Plan Appendix C instructions and examples on how to calculate the 10-year temporary increases (pages C-68 and -69) are of limited help because the two examples there apply the temporary access changes to the last years of the decade instead of EA Table 8's first years of the decade. Neither the Forest Plan, the Project EA or Exhibit Q-006 provide a reliable methodology that can be replicated in order to substantiate that the Betty Baptiste and Dry Riverside projects can occur simultaneously (due to Betty Baptiste being extended through 2025) without violating the 5-3-2 Standard (FW-STD-IFS-03). Nor does the EA provide any contingency scenario and assessment of effects should Dry Riverside be extended as well. Exhibit Q-006 does indicate "lower numerical impacts to OMRD, TMRD and CORE" if Dry Riverside were to start in 2026 rather than 2024, but the Proposed Action has not been adjusted to provide those lower impacts by not overlapping the two projects.

As we argued in our Objections to the revised Forest Plan, FW-STD-IFS-03 really is nothing more than an attempt to water down the true impacts of increased road access for logging and other projects. What the bears experience is an immediate and years-long impact from increased motorized access, not a "running 10-year average." The EA's treatment of FW-STD-IFS-03, as botched as it is, is a testament to why FW-STD-IFS-03's leniency to increased access and its associated logging impacts should not be allowed to stand.

That said, Exhibit Q-006 and the EA's reliance upon it is wholly inadequate. Nowhere are the road densities used in the calculations adjusted to reflect the existence of ineffective road closure devices, whether they be "persistent," known on occasion, or anticipated due to known ineffectiveness of closure methods, including methods for securing "impassable" roads. The calculations instead assume closures are 100% effective without providing any supporting inspection data and in the face of numerous court orders finding such an assumption is arbitrary, capricious and unlawful.

Culvert Replacements and Removals

All culvert removals must be considered essential mitigation under NEPA and must be fully funded before other Project actions can be taken, but they aren't (see 40 CFR 1505.3). Such crucial work, all of it in the Conservation Watershed Network in which "long-term conservation and preservation of bull trout and pure westslope cutthroat is prioritized" (EA at 102), must be implemented, not left to the vagaries of funding. Even the culvert work on haul routes is not guaranteed ("could be required", EA at 12).

The Plan BiOp requires that the Flathead "Remove all stream-aligned culverts when decommissioning roads in Conservation Watershed Network watersheds that have bull trout" (page II-78). "The project area is situated within the . . . Hungry Horse Reservoir Core Area for bull trout" (EA at 76). Not guaranteeing the removal of the 5 culverts on roads 5338, 1109 and 11410 as they are essentially decommissioned is a violation of the Plan BiOp's Terms and Conditions. Not guaranteeing the removal of these culverts is especially egregious given that these roads "have numerous culverts filled in with sediment or are prone to failure due to their locations on the landscape (avalanche chutes). (Scoping/PA document, parenthesis in original).

These roads "have limited benefits to the Forest's travel system" yet the EA claims "switching these roads from closed year-long barrier to impassable is an appropriate course of action" (EA at 12). This conclusion is arbitrary and capricious. The EA makes no mention of the requisite Travel Analysis and Report necessary to weigh the benefits and risks of either decommissioning these roads or retaining them in the System as impassable. Especially in the Conservation Watershed Network where bull trout are to be a priority, it is a no-brainer that these roads of "limited value [which] traverse numerous avalanche chutes and drainages which increase their maintenance needs" (EA at 12), should be decommissioned so that removal of the stream-aligned culverts is guaranteed and no further maintenance is needed per the Plan BiOp requirements. Simply rendering these roads impassable is an end-run around the Plan BiOp. The EA includes no monitoring and maintenance plan for these roads should the stream-aligned culverts not be removed or for the remaining cross-drain culverts and other drainage features – all of which will no longer be accessible by motor vehicles or equipment.

The Flathead has a long history of leaving stream-aligned culverts in abandoned and decommissioned roads, even though their removal was required by former Forest Plan Amendment 19 and common sense. This is evidenced by EA Table 45 (page 78) which says there are 146 culverts and bridges in the affected subwatersheds on "system **and historic roads** with culverts and bridges in place" (emphasis added). Table 3 (EA pages 8-9) makes clear that "Historical roads are roads that were NFS roads in the past but at some point, were removed from the NFS and decommissioned (as defined in the forest plan) . . . Existing template roads include roadbeds that are present on the ground but were never NFS roads."

This begs the questions: Why were culverts and bridges left in decommissioned historic roads and how many of them were also left in "existing template roads? The EA is silent on this issue and provides no listing we can find of how many culverts and bridges remain in “existing template roads.” Instead, the EA continues the Flathead’s bad habits of leaving culverts in impassable roads while avoiding the decommissioning of roads so it need not remove culverts from them. The Flathead tried to correct its bad habits through Amendment 19 but reneged on it in its revised Forest Plan, demonstrating it cannot be rehabilitated in light of its desire to build and retain as much road access as possible while scuttling adequate protections to fish and wildlife.

Oddly, EA Tables 45 - 49, among others, include “Quintonkon Creek – Hungry Horse Reservoir” among the four HUC-12 subwatersheds affected by the Project. Why is this subwatershed, which is across the Reservoir from the Project area included? Table 48 then shows the Project will result in 4 additional stream crossings on the east side of the Reservoir, but will have 4 less crossing on the west side of the Reservoir, as though the Project will have no net impact. How does this Project result in 4 less stream crossings in the Quintonkon subwatershed outside the Project area, or is Quintonkon included only to give the appearance of no net increase in stream crossings?

Climate Impacts

The EA provides a single paragraph (page 101) on the effects of Dry Riverside on forest carbon cycling. It simply tiers to the Forest Plan and references Project File Exhibit R-29, concluding “The proposed action will not convert forestland to other non-forest uses. Any carbon initially emitted from the proposed action will have a temporary influence on atmospheric CO2 concentrations as carbon will be removed from the atmosphere over time as the forest regrows or recovers.” How long will that take? This is not the “hard look” required by NEPA, even when inclusive of the cited Exhibit R-19 and Forest Plan pages.

In a nutshell, the cited Forest Plan FEIS pages (288-311) provide a wholly inadequate and biased accounting of the forest carbon cycle. By claiming that “Carbon stored in harvested wood products contributes to the total forest carbon storage” is misleading because only a small percentage of the carbon removed as sawlogs actually becomes a wood product while the rest is wasted or burned as biomass. Trying to compensate for this waste by claiming landfills are some “of the fastest-growing carbon pools” is equally ludicrous. It is also equally misleading to claim “when the effect of substituting wood for concrete and steel was also accounted for, then harvest scenarios resulted in less CO2 emission than the no-harvest scenario.”

Neither the Project EA or the Plan FEIS disclose to what degree wood is replacing concrete and steel in the area where the wood products are being produced. Neither of them discloses what percentage of the carbon removed during logging is actually turned into a wood product, how much is burned on-site as slash, how much ends up in a landfill, and how much is burned as biomass fuel, such as in Stoltze’s Columbia Falls sawmill. Both essentially claim that cutting down an old tree and putting it in the landfill releases less CO2 than allowing that tree to continue living and sequestering CO2, then dying and continuing to sequester CO2 while also replenishing soil and duff.

Indeed, Smith et al (2006) and Gower et al (2006) show that as little as a net 15% of harvested wood ends up as stored carbon when logging residue, mill residue and transportation emissions are accounted for. See the attached "U.S. Forest Carbon and Climate Change" report by The Wilderness Society, at 11.

Moreover, Campbell et al (2007) show that only some 5% of a tree's carbon is released during a wildfire because the bole of the tree does not usually burn. This is the part of the tree targeted for removal as logs during logging, which removes that carbon from the forest ecosystem. The Campbell paper (attached) show that the majority of carbon released to the atmosphere during wildfire is from the litter and duff.

The photo below is of a slash pile left to be burned in the Flathead's Bug Creek Project and was taken 8/26/23 from Road 498. Note the large amount of tree boles that would likely not have burned in a wildfire, but were nonetheless cut down and left as slash to be burned and have the carbon released into the atmosphere. This is not an isolated incidence or practice.

The photo below is of a slash pile left to be burned in the Flathead's Middle Fork Fuels Reduction Project and was taken 9/3/17 from Road 1637. Note the large amount of tree boles that would likely not have burned in a wildfire, but were nonetheless cut down and left as slash to be burned and have the carbon released into the atmosphere.

The Project EA, Plan FEIS and Exhibit R-19 all fail to adequately quantify the effects of the Project on the forest carbon cycle and the climate, while also failing to square with the actual logging practices on the Flathead NF. Rather than account for the climate impacts of logging, they discount the impacts of logging by comparing to other forest carbon-releasing events such as fire, insects and disease. This even though logging on the Flathead, by its own estimation, is nonetheless estimated to account for as much as 10% of the negative effect on carbon storage (Plan FEIS at 310), **while totally ignoring the fact that logging is the one carbon-releasing event that the Forest Service has absolute control over!**

In the end, the FEIS concludes "timber harvest would have little impact overall on the potential future scenario of carbon accumulation and loss" and the EA simply repeats this conclusion in Exhibit R-19 and by tiering to the FEIS. This lame approach was recently faulted by the U.S. District Court in Missoula in its decision on the Kootenai National Forest's Black Ram Project, writing "while the USFS did address climate change in its review, merely discussing carbon impacts and concluding that they will be minor does not equate to a 'hard look' [as required by NEPA]." (*Center for Biological Diversity v. US Forest Service*, CV 22-14-M-DWM, 8/17/23).

Need for Broad Range of Alternatives and an EIS

The EA has an inadequate range of alternatives, being limited to the No Action and Proposed Action alternatives only. Where is an alternative that would meet the 19/19/68 research benchmarks for OMRD/TMRD/CORE that would avoid incidental take of grizzly bear, as described in the Plan BiOp at III-97, -98? Where is an alternative that would describe the various effects of NOT having the Betty Baptiste and Dry Riverside projects overlap in time? These are reasonable alternatives. NEPA requires a broad range of reasonable alternatives to compare to each other and the No Action alternative, not just a single Proposed Action alternative.

The EA essentially admits that substantial uncertainty exists regarding the effects of the proposed action. On page 7, the EA states "Several commentors expressed concern and provided literature suggesting that the proposed vegetative treatments may increase fire behavior . . . Additional references were added [by the FS]to the project file to further document that fuel treatments in general do reduce the negative outcomes of wildfire." Discussion of significant disagreement and uncertainty regarding the effects of the proposed action must be conducted in an EIS, not an EA, according to NEPA and its implementing regulations. We have also demonstrated in these comments and in our Road Hunt report that substantial uncertainty exists regarding the effectiveness of road closures, which also requires the preparation of an EIS. We have also shown that substantial uncertainty exists regarding the effects of the proposed action on climate change, which also requires the preparation of an EIS.

Thank you for this opportunity to comment.

Sincerely,

Keith

Keith J. Hammer
Chair

Attachments:

1. 2007 U.S. Forest Carbon and Climate Change report by The Wilderness Society

2. Campbell et al (2007)

Provided separately:

3. "Road Hunt: A Survey of Road Closure Effectiveness," (Hammer 2023)

ATTACHMENT 1

U.S. Forest Carbon and Climate Change

Controversies and Win-Win Policy Approaches

Economic Analysis

SCIENCE FROM

THE WILDERNESS SOCIETY

Our Mission

Since 1935, **The Wilderness Society** has worked to preserve America's unparalleled wildland heritage and the vast storehouse of resources these lands provide. From the threatened tupelo and cypress forests of the Southeast to critical grizzly bear and wolf habitat in the Yellowstone-to-Yukon corridor to the incomparable, biologically rich Arctic, The Wilderness Society has forged powerful partnerships with members and friends across the country to conserve interconnected landscapes for our nation. We want to leave a legacy rich in the biological diversity and natural systems that nurture both wildlife and humans alike.

Headquartered in Washington, D.C., the Society also maintains nine regional offices where our staff address on-the-ground conservation issues linked to local communities. Since spearheading passage of the seminal Wilderness Act in 1964, we have been a leading advocate for every major piece of Wilderness legislation enacted by Congress, work that is supported by an active membership of more than 200,000 committed conservationists. Our effectiveness stems from a team approach to conservation, which links our scientists, policy experts, and media specialists to thousands of grassroots activists — creating a potent force to promote change.

Building the case for land preservation with tactical research and sound science is the key to successful environmental advocacy and policy work. Nearly a quarter century ago, The Wilderness Society helped pioneer strategies that incorporated expert economic and ecological analysis into conservation work. Today, through focused studies, state-of-the-art landscape analysis — and diligent legwork by our many partners who provide us with on-site data — our **Ecology and Economics Research Department** is able to serve the needs of the larger conservation community.

Legislators, on-the-ground resource managers, news reporters, our conservation partners, and — most importantly — the American people must have the facts if they are going to make informed decisions about the future of this nation's vanishing wildlands. The answers to the pressing legal, economic, social, and ecological questions now at issue are the stepping stones to that understanding and, ultimately, to achieving lasting protection for the irreplaceable lands and waters that sustain our lives and spirits.

ATTACHMENT 1

U.S. Forest Carbon and Climate Change

Controversies and Win-Win Policy Approaches

by
Ann Ingerson

THE WILDERNESS SOCIETY

ATTACHMENT 1

Acknowledgments

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And finally, thanks to Sarah DeWeerdt and Mitchelle Stephenson who with infinite patience, attention to detail, and efficiency molded this report into its final form.

Citation

Ingerson, Ann L. 2007. U.S. *Forest Carbon and Climate Change.* Washington, D.C.: The Wilderness Society.

This science report is one of a series that stems from conservation research studies conducted by The Wilderness Society's Ecology and Economics Research Department. Other reports and briefs in the series that focus on issues relevant to this report include:

- **Environmental Benefits and Consequences of Biofuel Development in the United States:** a Science and Policy Brief
- **Fragmenting Our Lands:** The Ecological Footprint from Oil and Gas Development (A Spatial Analysis of a Wyoming Gas Field)
- **Energy & Western Wildlands:** A GIS Analysis of Economically Recoverable Oil and Gas
- **Wildlife at a Crossroads:** Energy Development in Western Wyoming

These and other reports and briefs are available on The Wilderness Society's web site <www.wilderness.org> and from The Wilderness Society, Communications Department, 1615 M Street, NW, Washington, DC 20036 (202-833-2300 or 1-800-THE-WILD).

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Preface

The United States is blessed with a rich tapestry of forested landscapes—from the shade-dappled hardwood stands of New England to the open pinelands of the Southeast and towering firs of the Pacific Northwest coast. Woodland habitats shelter thousands of wildlife species and provide a treasure trove of recreation opportunities for the American people. In addition, our forests store vast amounts of carbon in tree trunks, roots, leaves, dead wood, and soils—a service that is becoming ever more essential as the threat of global climate change mounts due to the buildup of human-generated carbon dioxide and other greenhouse gases in the atmosphere.

Although investments in energy efficiency and clean energy will provide the only permanent solutions to climate change, forest sequestration can buy us time to develop those alternatives. U.S. forests currently capture the equivalent of about one-tenth of the nation's greenhouse gas emissions. They have the potential to contribute even more to climate change mitigation. But this potential will only be realized if we move carefully, with properly designed policies to increase forest carbon stores.

The Wilderness Society's report, *U.S. Forest Carbon and Climate Change*, examines various policy options to promote the role of forests in carbon sequestration. After a thorough review of the available data measuring and accounting for the amount of carbon stored in and moving through forest ecosystems, author Ann Ingerson presents persuasive evidence about the challenges inherent in many current proposals. Some frequently discussed solutions are much more complex than they first appear. Others such as carbon markets, for example, may present risks around the issues of permanence and measurement, which could hamper their effectiveness as tools for meeting the climate challenge long-term. Several strategies, if adopted without careful consideration of their full carbon-cycle effects, could actually *decrease* the amount of carbon stored in our forests.

Fortunately, several simple and broadly supported policy approaches for increasing forest carbon stores also exist. Protecting the forests we have, replanting depleted landscapes, and managing forests for longer rotations and larger volumes of standing timber will all help ensure these critical wildlands play an ongoing role in climate change mitigation. A host of related benefits will accrue from such policies, including habitat for species, recreation opportunities, and key public values such as water filtration. One way to begin to address the global warming issue is to look to these strategies first to increase forest carbon stores. This approach may also provide the vehicle for bringing together some unusual allies—from environmental NGOs to private forestland owners and the wood products industry—ready to find common solutions to the climate problem that threatens us all.

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Acronyms and Abbreviations

CO_2e	Carbon dioxide equivalent
DOE	U.S. Department of Energy
EPA GHG	U.S. Environmental Protection Agency Greenhouse Gas Inventory
FIA	Forest Inventory and Analysis (program of the U.S. Forest Service)
MMT	Million metric tons (teragrams)
NGO	Non-governmental organization
NRI	National Resources Inventory (program of the U.S. Department of Agriculture)
USDA	U.S. Department of Agriculture

ATTACHMENT 1

Executive Summary

As consensus grows about the serious impacts of global climate change, the role of forests in carbon storage is increasingly recognized. Terrestrial vegetation worldwide currently removes about 24 percent of the greenhouse gases released by industrial processes. Unfortunately, this contribution is approximately cancelled out by carbon released as a result of global deforestation and other ecosystem changes. Slowing or halting the rate of deforestation is thus one of the prime strategies to mitigate global climate change.

The U.S. situation differs from the global one in several ways. Since both forest acres and average biomass per forest acre are currently increasing, as U.S. forests recover from past clearing or heavy harvest, our forest carbon stores are growing larger over time. However, our high rate of industrial emissions means that only about 10 percent of the carbon released from burning fossil fuels in the United States is captured by our forests. Moreover, net U.S. forest carbon sequestration has begun to slow in recent years as reforestation reaches its limits and development sprawls into more rural forested areas. U.S. forests could possibly capture a much higher portion of our industrial emissions, but only if we prevent forest conversion and development and manage our forests to maximize carbon stores.

How can we develop effective policies to protect and enhance forest carbon stores? A first step is to understand the magnitude of carbon emissions and storage. International discussions about global climate change have led governments at national and state levels to document greenhouse gas emissions and stores through economy-wide inventories or voluntary registries, most of which include special provisions for the forest sector. The next step would be to enact policies that encourage increased forest sequestration. Widely publicized carbon markets under the Kyoto Protocol have tended to focus policy discussions rather narrowly on the sale of forest-based carbon offsets to greenhouse gas emitters under a cap-and-trade scheme. But before forest-based offsets can become a tradeable commodity, several issues need to be addressed, including the need for a consistent and verifiable accounting system, the need to prove additionality over some well-defined baseline, and the need to guarantee permanence of carbon storage.

Given the uncertainties about offsets as a tradeable commodity, other public policies to enhance forest carbon stores may be a better option. One approach might be to maintain a large carbon bank on public forestland; another would be to subsidize private landowners who increase carbon storage on their forestland.

Whether we use marketable offsets or other public policies as tools, managing forest carbon to mitigate climate change is a complex business that requires understanding the entire carbon cycle over long time periods. Three strategies often proposed as forest-based climate change solutions illustrate some of these underlying complexities:

1) **Does replacement of old, slow-growing forests with young, intensively managed plantations speed carbon sequestration?** Since net biomass growth rates slow down in mature forests, keeping forests in a young, fast-growing state through

In this report, we explore:

1. The role of forests in sequestering carbon dioxide—thus mitigating global climate change—and the state of the U.S. forest carbon bank account.
2. The complexities of measuring forest carbon, particularly using such tools as inventories and registries.
3. Some potential pitfalls of cap-and-trade programs, markets for forest-based carbon offsets, and subsidies to boost forest carbon.
4. The complexities of three specific forest-based strategies often proposed for mitigating climate change: managing for fast-growing young forests, increasing carbon stored in wood products, and increasing use of woody biomass fuels.
5. Policy approaches to boosting forest carbon that have many secondary benefits for the public and the environment as well: forest preservation, restoration, and sustainable management.

short-rotation harvests would seem a reasonable strategy for enhancing carbon sequestration. However, only a full accounting will determine whether a regenerating forest fixes more carbon than the mature forest it replaces. Rather than simply comparing live-tree carbon fixed annually by old and young trees, we need to compare *all* carbon flows over time for a mature forest (including accumulations in dead woody biomass and soil) to *all* flows associated with a harvested forest (including harvest-related emissions and wood products carbon losses).

2) **Does converting trees into long-lived wood products increase carbon stores?** Forestland owners would like to claim credit for carbon harvested and stored off-site in long-lived wood products. Though intuitively appealing, this approach presents several unresolved questions, including how to account for emissions related to harvest and processing, the uncertainty of permanent stores not controlled by the landowner, and how to credit emissions reductions due to substituting wood for other building materials. With multiple decision-makers dispersed throughout the national and even global marketplace, tracking the fate of harvested carbon is a challenge.
3) **Is woody biomass a carbon-neutral fuel?** It is often argued that woody biomass sequesters as much carbon while growing as it releases when burned, and hence should be eligible for offset credits when it replaces fossil fuel use. To assure carbon neutrality, however, the source forest must be protected from conversion and managed so as to replace all carbon released by burning. Even with such management, energy conversion losses and emissions from harvest, transport, and chipping will pull the ratio of carbon fixed to carbon released below 1:1.

As we work to better understand the long-term carbon impacts of forest management decisions, it makes good sense to start with strategies for increasing forest carbon that also provide secondary public benefits. Forest preservation and reforestation maintain or increase forested area, and also provide habitat for forest-dependent species, improve water quality, and regulate floodwaters that may become more severe as the climate changes. Lengthening rotations and increasing standing timber volumes enhance scarce late-successional habitat, provide more high-quality timber, and create forest surroundings that are attractive for remote hiking, fishing, and other back-country recreation. Beginning with these low-risk approaches will help achieve consensus about the contributions of forests to moderating climate change and build support for public policies that protect and enhance their role.

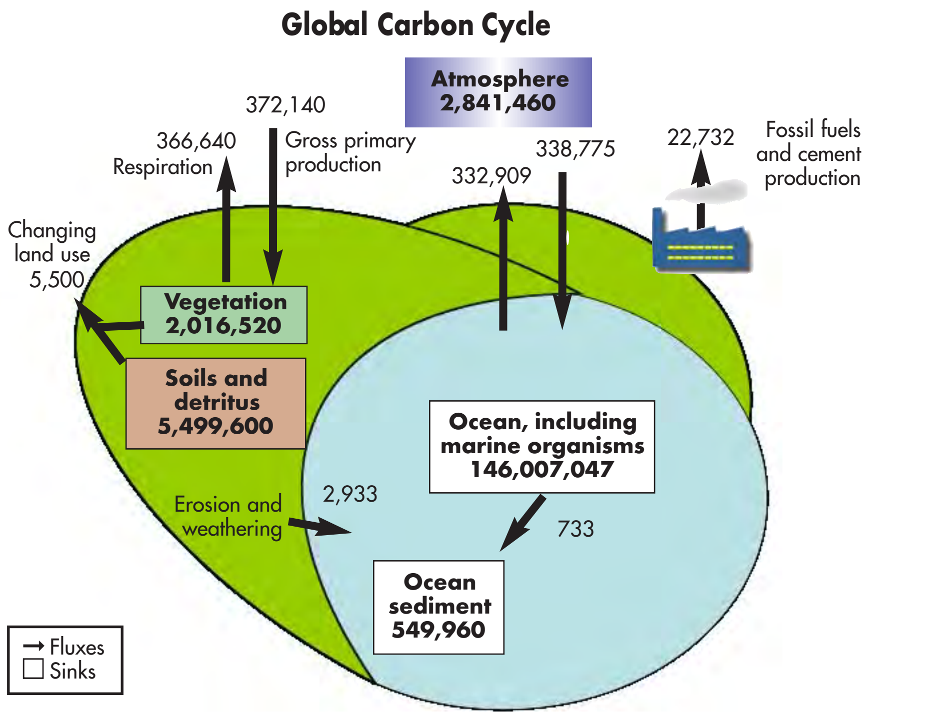
Forests and the Global Carbon Cycle

Societies around the globe are beginning to address the threat of severe climate change through policies aimed at reducing the buildup of greenhouse gases. Natural ecosystems, including forests, are a critical link in the global carbon cycle and must play a vital role in the mitigation of global warming. Forests are important both for their large *existing reservoirs* of carbon (often called "pools" or "sinks") and because of the *ongoing net flow* of carbon from the atmosphere into that forest reservoir (often called "flux"). Figure 1 shows the major global sources, sinks, and annual fluxes of carbon.

Currently, land-based stores of carbon dioxide equivalent[1] are about 7,516,120 million metric tons (MMT) worldwide. This carbon bank account is continuously built up or depleted by photosynthesis, respiration, and erosion, and also through restoration, destruction, or change of various landscape types. For all lands that support plant growth (forests, croplands, wetlands, etc.), the carbon dioxide removed from the atmosphere by photosynthesis—372,140 MMT/year—generally exceeds that released through respiration by plants and decomposer organisms—366,640 MMT/year—meaning that growing plants and associated fungi and bacteria remove a net 5,500 MMT of carbon dioxide from the atmosphere each year (about 24 percent of the carbon released by industrial processes).

Photosynthesis will continue to exceed respiration overall, however, only with proper management of existing landscapes. Clearcutting a forest, for instance, boosts respiration (releasing CO_2) and suppresses photosynthesis (reducing biological fixation of CO_2) for several years or decades—even when land is replanted or allowed to regenerate

FIGURE 1.



All figures given in millions of metric tons CO_2e.

Adapted with permission from Oak Ridge National Laboratory. Source figure available from http://cdiac.ornl.gov/pns/graphics/c_cycle.htm. 1992-1997 data.

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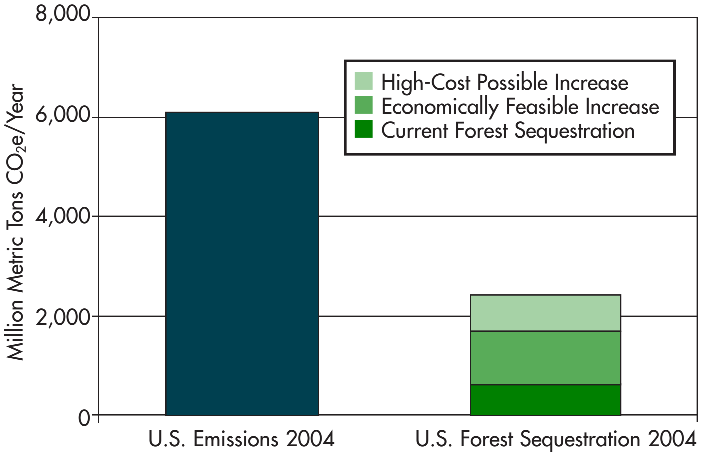
[1] Carbon budgets can be confusing because of the variety of units utilized. Millions of metric tons (teragrams) is fast becoming the standard unit of measurement, but some sources report the mass of elemental carbon stored, while others use the mass of CO_2 (3.6664 times the mass of C) or include all greenhouse gases as CO_2 equivalents (often abbreviated CO_2e). This last unit is important because, though CO_2 is the main gas responsible for global warming, other gases make an even greater contribution to the greenhouse effect. Methane (CH_4), for instance, is about 21 times as potent as CO_2 pound-for-pound and over time, and N_2O is 310 times as potent. In order to gauge the capacity of forests to offset emissions, we will express carbon quantities in CO_2e (primarily millions of metric tons) through the rest of this paper.

FIGURE 2.
U.S. Industrial Greenhouse Gas Emissions and Current and Potential Forest Sequestration



Emissions and sequestration data from U.S. Environmental Protection Agency 2007. Economically feasible and high-cost possible forest sequestration from Stavins and Richards 2005.

trees. Large existing stores of carbon are released into the atmosphere when land is converted to other uses. Since more land is developed, drained, or otherwise converted annually than is restored to its natural cover, land use changes release about 5,500 MMT of CO_2 each year, essentially negating the entire contribution of plants to the land-based carbon sink.

U.S. Forests as Carbon Sinks

U.S. forests store about 152,236 MMT CO_2e, representing about 2 percent of global terrestrial carbon stores. An additional 8,781 MMT CO_2e are stored in wood products in use and in landfills (U.S. Environmental Protection Agency 2007). Though deforestation is occurring much more rapidly than forest growth globally, forests in the United States currently remove substantially more carbon from the atmosphere than they emit, so our forest-related carbon sink is increasing by about 699 MMT CO_2e annually (a growth rate of 0.4 percent).[2] In the eastern United States, land formerly cleared for farming is growing back naturally to woods or is being replanted through conservation assistance programs like the USDA Conservation Reserve Program. In the Pacific Northwest, forestlands are recovering from intensive harvesting during the mid-to-late 20th century, and are rebuilding large carbon stores in the form of living trees above and below ground, shrubs, snags and coarse woody debris, soil, and forest floor litter.

The United States, with 4 percent of the world's population, is responsible for nearly one-quarter of global carbon emissions. As our nation develops a long-overdue strategy to reduce our climate change impact, we must protect our existing stores of forest carbon and also enhance the capacity of our forests to fix additional carbon in the future. Figure 2 compares estimated annual U.S. industrial emissions of greenhouse gases with net annual carbon sequestration by U.S. forests. Our forests currently sequester about 10 percent of U.S. industrial emissions of CO_2-equivalent gases; given the right policies that proportion could reach as high as 36 percent, though high costs make it unlikely we will ever reach that goal. Although investments in energy efficiency and clean energy will provide the only permanent solutions to climate change, forest sequestration can buy us time to develop those alternatives. Relatively low-cost policies to increase forest carbon stores include protection of existing forestland from development, restoration of deforested or degraded lands, and management to increase carbon stores on existing forestland.

An Uncertain Future for U.S. Forest Carbon Stores

Though U.S. forests currently help offset our industrial carbon emissions and could potentially contribute even more, the ability of our forests to continue providing this important service is in question. Our total stores of forest carbon are still increasing each year, but at an ever-slower rate. Figure 3 shows historic carbon fluxes to and from forests in what is

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[2] Since the increase in our forest carbon sink is based solely on the difference between starting and ending inventory, it does not reflect the contribution of woody biomass replacement of fossil fuels to reducing greenhouse gas emissions.

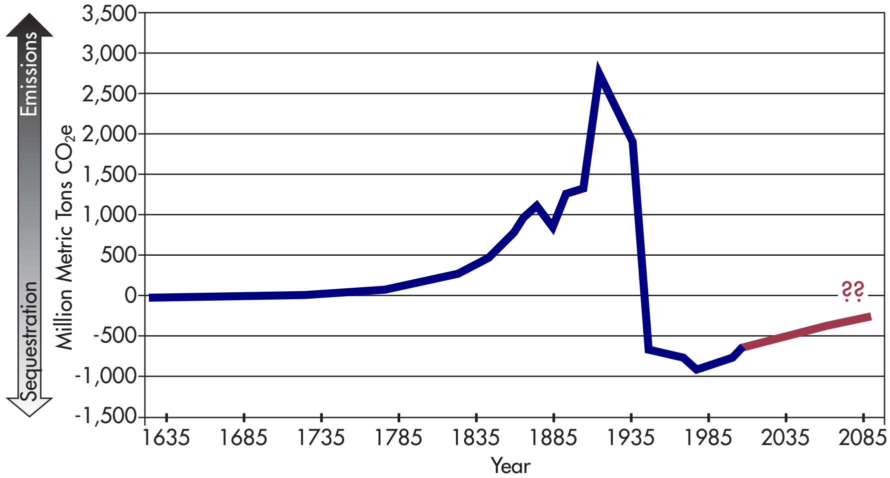
now the United States (including both the forested ecosystem and the carbon derived from it but stored off-site in wood products). Note that positive numbers in the figure represent emissions, and negative numbers represent sequestration. European settlement and accompanying deforestation made our forests net sources of carbon emissions by the mid-1700s, a trend that peaked in the early 1900s. By the mid-1900s, regrowth of forests on abandoned farmland and cut-over timberlands began to replenish our national carbon bank account. In recent years, however, net annual flows of carbon out of the atmosphere and into the forest ecosystem and wood products pools have begun to decline once more. If recent trends continue (red line), our forests may cease to sequester net carbon by the end of this century.

Forest carbon stores are threatened by both reduced forest acreage and reduced carbon density (tons of carbon stored per acre). The U.S. Forest Service's Forest Inventory and Analysis (FIA) Program provides information about trends in forest acreage. Though FIA data show gains in forest acreage for the United States as a whole in recent years, these gains are not uniform and in fact 23 of the 48 coterminous U.S. states lost forest acreage between 1997 and 2002 (Figure 4).

There is much uncertainty regarding the accuracy of these acreage figures, which are derived from periodic sampling and suffer from occasional changes in the definition of forestland. For example, some of the data on which calculations of forestland losses for 1997-2002 are based were collected as far back as the early 1990s, and probably fail to accurately reflect recent changes in forestland acres. Data are also from samples rather than complete land cover analysis, and sampling errors are relatively high. However, these are the best data currently available on a nationwide basis. Efforts are underway to improve estimates of forest area changes.

FIGURE 3.

U.S. Forest Carbon Budget (Ecosystem + Wood Products)



Adapted with permission from Birdsey et al. 2006.

Gross acreage changes also mask the fact that acreage gains often apply to early regrowth of abandoned farmland that is severely depleted in carbon stores, while losses may occur in high-carbon mature forests at the suburban sprawl frontier. The U.S. Department of Agriculture's National Resources Inventory (NRI) allows us to track conversion between specific land cover types (U.S. Department of Agriculture 2000). Though recent changes cannot yet be assessed due to a change in sampling methods, NRI data indicate a net increase of 3.6 million acres of forestland nationwide from 1982 to 1997. Over this period more than 8 million acres of forest were converted to agricultural uses and 12 million acres were developed or converted to "other rural land," while 23 million acres of new forest began to grow on former farmland. Overall, this exchange of acres would cause a net loss of forest carbon.

Estimates of carbon released through land conversion vary widely, as some kinds of low-density development may keep forests nearly intact. But many sources agree that carbon losses due to forest conversion are significant. The Pacific Forest Trust (Gordon 2006) estimates that "probably, upwards of 25 tons

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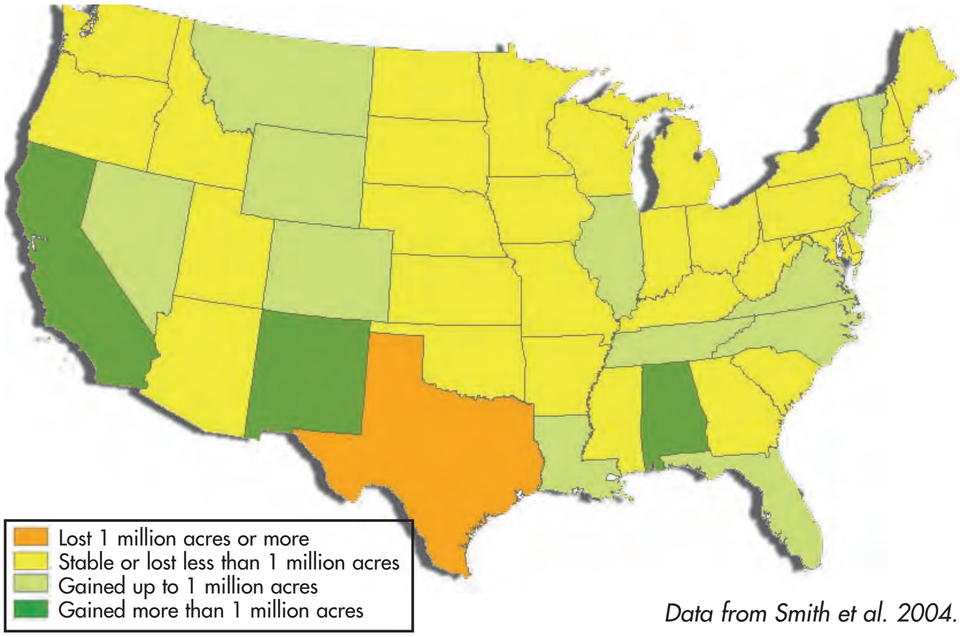
FIGURE 4.

Estimated Change in Forestland Area, 1997-2002



Data from Smith et al. 2004.

of carbon emission per acre [83 metric tons CO_2e] can be prevented for each acre not converted from forest to another use," and that 1.5 million acres of forest lost every year to development in the United States release 275 million metric tons of CO_2e (Pacific Forest Trust 2007). In the Northeast, roughly 150 tons of CO_2e are released for every forested acre developed.[3] Moreover, when forestland is converted to other uses, not only is CO_2 released but the land's future capacity to continue drawing carbon dioxide out of the air may be diminished or lost.

[3] According to the North East State Foresters Association (2002), the forests of New York and New England contain, on average, 106 metric tons of total carbon (388 metric tons CO_2e) per acre, with about one-third in live trees. Environment Northeast (Stoddard and Murrow 2006) estimates that 50-67 percent of above-ground carbon and 22-25 percent of soil carbon are released on conversion. Putting these figures together yields 139 to 178 metric tons CO_2e emitted per acre converted in the Northeast.

Measuring Forest Carbon

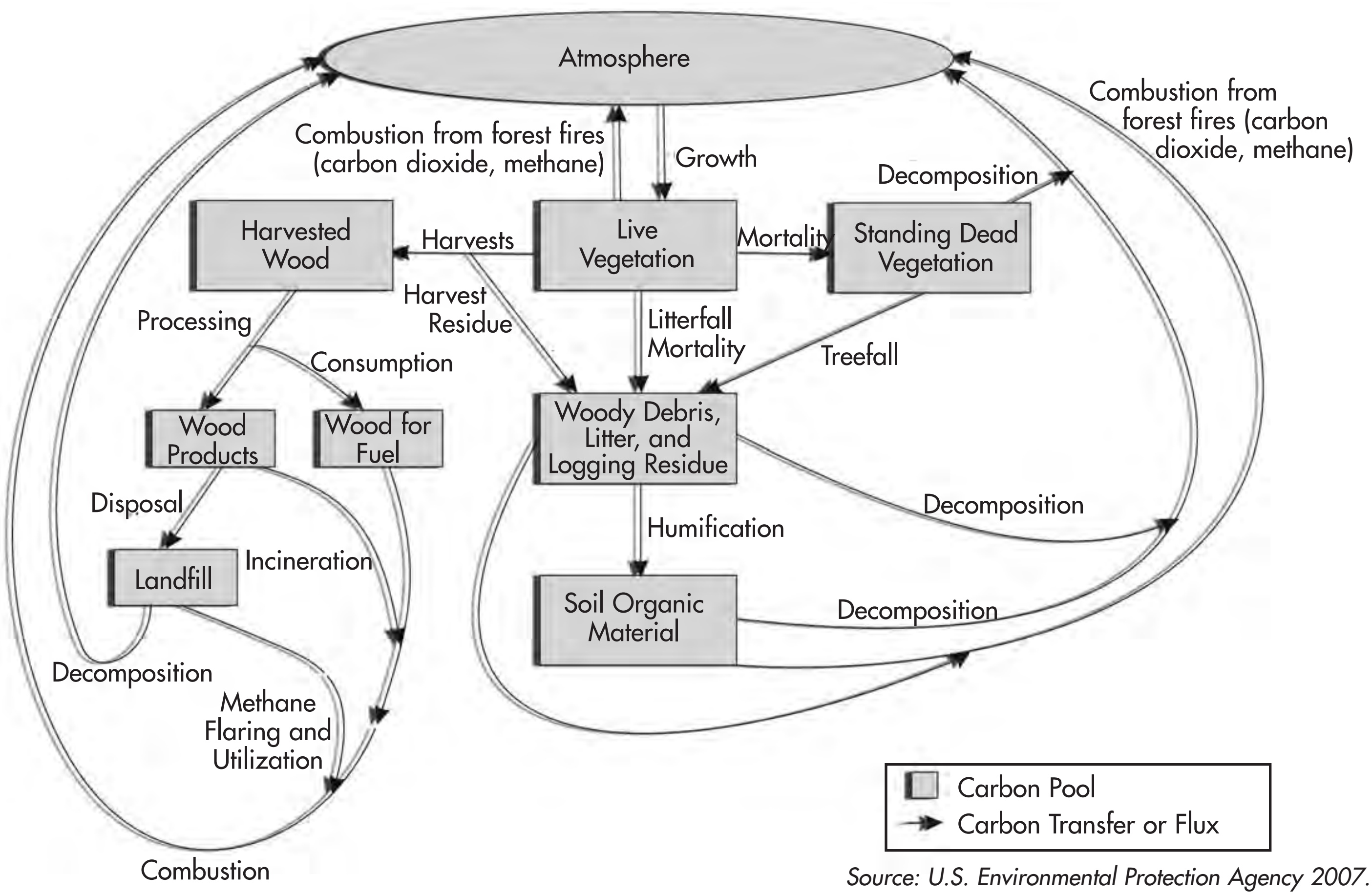
Protecting and enhancing forest carbon is an effective way to reduce greenhouse gases, but its use as a public policy tool will require careful documentation. Official national inventories and voluntary registries at national and state levels are designed to track carbon stores and changes in those stores. A brief look at these tools shows that our capacity to measure all pools of carbon associated with forests is very limited, and we need much better information to manage this resource to its full potential.

The official national inventory of carbon stocks (pools) and average annual changes (fluxes) in greenhouse gases across the entire U.S. economy is the Environmental Protection Agency's annual Greenhouse Gas Inventory (EPA GHG). Policymakers turn to this comprehensive national record to assess U.S. contributions to climate change and will use it in the future to evaluate the effectiveness of mitigation measures. The USDA Forest Service is tasked with developing forest carbon numbers for the Land Use and Land Use Change segment of this inventory. Figure 5, developed by Linda Heath of the USDA Forest Service, illustrates the complexity of tracking forest carbon. Table 1 shows the most recent EPA GHG estimates of changes in forest carbon stores in the United States.

Most of the data in the EPA GHG Inventory comes from the Forest Inventory and Analysis Program. The FIA provides the only nationwide infor-

FIGURE 5.

Forest Sector Carbon Pools and Fluxes



Source: U.S. Environmental Protection Agency 2007.

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TABLE 1.

EPA Greenhouse Gas Inventory Estimates of Changes in Forest Carbon Stores

Carbon Pool	1990	1995	2000	2001	2002	2003	2004	2005
Forest	**(466.5)**	**(602.0)**	**(529.4)**	**(555.5)**	**(595.3)**	**(595.3)**	**(595.3)**	**(595.3)**
Aboveground Biomass	(251.8)	(331.0)	(347.1)	(360.4)	(376.4)	(376.4)	(376.4)	(376.4)
Belowground Biomass	(63.9)	(69.8)	(73.9)	(76.4)	(79.5)	(79.5)	(79.5)	(79.5)
Dead Wood	(36.7)	(60.9)	(48.2)	(50.0)	(52.4)	(52.4)	(52.4)	(52.4)
Litter	(65.6)	(49.5)	(35.8)	(47.1)	(52.2)	(52.2)	(52.2)	(52.2)
Soil Organic Carbon	(48.5)	(90.8)	(24.5)	(21.6)	(34.8)	(34.8)	(34.8)	(34.8)
Harvested Wood	**(132.0)**	**(115.5)**	**(109.3)**	**(90.2)**	**(92.8)**	**(91.7)**	**(102.0)**	**(103.4)**
Wood Products	(63.1)	(53.5)	(46.2)	(31.2)	(34.1)	(33.4)	(43.3)	(44.4)
Landfilled Wood	(68.9)	(62.0)	(63.1)	(59.0)	(58.7)	(58.3)	(58.7)	(59.0)
Total Net Flux	**(598.5)**	**(717.5)**	**(638.7)**	**(645.7)**	**(688.1)**	**(687.0)**	**(697.3)**	**(698.7)**

Note: All figures given in units of MMT CO_2. Forest C stocks do not include forest stocks in Alaska, Hawaii, or U.S. territories, or trees on non-forest land (e.g., urban trees, agroforestry systems). Parentheses indicate net C sequestration (i.e., a net removal of C from the atmosphere). Total net flux is an estimate of the actual net flux between the total forest C pool and the atmosphere. Harvested wood estimates are based on results from annual surveys and models. Totals may not sum due to independent rounding.

Source: U.S. Environmental Protection Agency 2007.

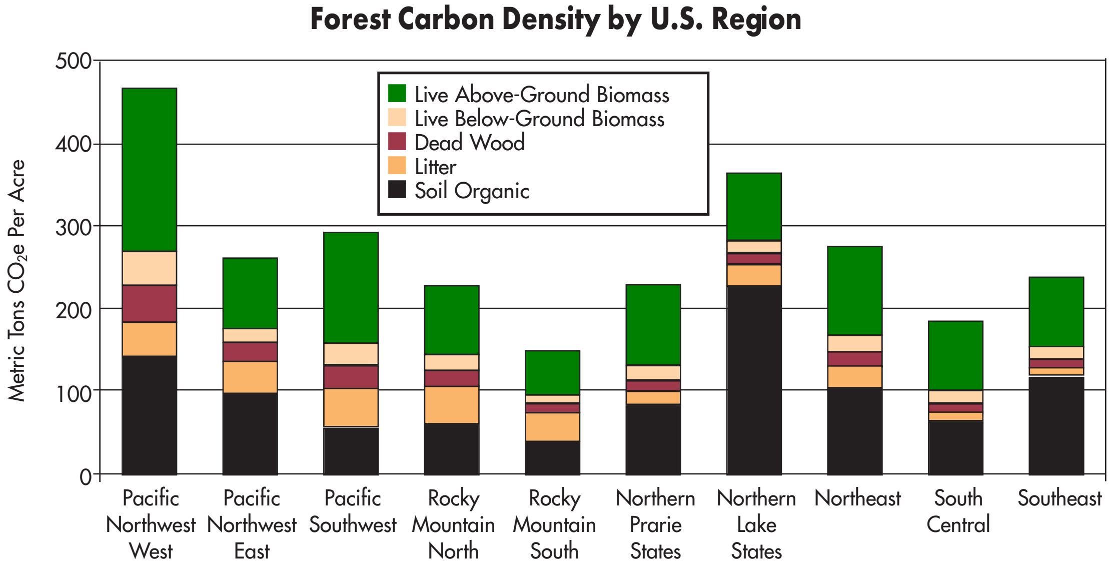
mation about forest resources over time, and it was originally designed to track commercial timber resources, not to measure carbon. As a result FIA data suffers from many limitations (though plans are underway to address most of them if funding permits):

- FIA has only recently begun to measure biomass, forest floor debris, and other variables important for assessing carbon stocks. Soil carbon is not monitored and so estimates are based on broad forest types regardless of land use history.
- FIA inventories for some states are 15 to 20 years old and early sampling protocols varied from state to state. Lack of frequent updates forces researchers to interpolate between sampling dates, resulting in anomalies like the constant forest data for 2002 through 2005 in Table 1.
- Limited inventory data for Alaska means that important state is excluded altogether.
- The EPA GHG Inventory excludes altogether any measures of the impact of development and land use change on forest carbon stores, citing a lack of adequate data on land use changes.

Figure 6 illustrates why the lack of information about soil organic matter, dead wood, and litter might matter. These nonliving components make up a substantial fraction of total forest carbon in all regions—from a low of 45 percent in the Pacific Southwest to a high of 73 percent in the Northern Lake States. These are the ecosystem components that tend to be most depleted under intensive management, particularly in forests regenerating from cleared agricultural lands. Managing forests to restore natural levels of these components could yield substantial carbon sequestration benefits.

In addition to the nationally aggregated EPA GHG *inventory*, another compendium of information on forest carbon stocks is the U.S. Department of Energy's voluntary *registry* that allows individual entities to report their own emissions and sequestration of greenhouse gases. This national registry is often called 1605(b) for the section of the Energy Policy Act of 1992 that required its

FIGURE 6.



Regions: PNWW (Western OR and WA); PNWE (Eastern OR and WA); PSW (CA); RMN (ID, MT); RMS (AZ, CO, NM, NV, UT, WY); NPS (IA, IL, IN, KS, MO, ND, NE, SD); NLS (MI, MN, WI); NE (CT, DE, MA, MD, ME, NH, NJ, NY, OH, PA, RI, VT, WV); SC (AL, AR, KY, LA, MS, OK, TN, TX); SE (FL, GA, NC, SC, VA).

Data from Smith and Heath 2006.

establishment. Some states and several private organizations have also developed registries, each with its own system of accounting for carbon stores, emissions, and sequestration. For example, registries may differ in:

- Reporting by entity versus by project (a single tree planting project may be undercut by increased timber cutting by the same company elsewhere).
- Which carbon pools must be measured (increases in wood products carbon might eventually result in depleted soil carbon pools).
- Method of monitoring (models or look-up tables may be less reliable, but also more affordable, than on-the-ground sampling).

Registry standards determine to what extent a forestland owner or a forest sequestration project can claim credit for mitigating climate change. Therefore, establishing a uniform method of accounting is key to making registries work in the future.

Policies to Protect and Enhance Forest Carbon

Mitigating climate change is a classic public good, with benefits that are non-exclusive (if one person benefits, we all do) and non-competitive (one person's enjoyment of a more natural climate regime in no way diminishes others' enjoyment of the same). Policy mechanisms to provide public goods can be either market-based or government-run, or some combination of the two. In the case of greenhouse gas reductions, market solutions in the form of cap-and-trade mechanisms have received much attention, due to their prominent role in the Kyoto Protocol. However, trading of forest-based carbon offsets presents several challenges, and other policy alternatives should also be considered.

Cap-and-Trade Programs and Offsets

Cap-and-trade is a flexible regulatory tool in which a maximum emissions allowance (cap) is set for regulated sources of greenhouse gases. The system then allows those sources to meet their cap either by reducing their own emissions, or by purchasing excess reductions or carbon sequestration offsets from others (trade). Marketed forest-based offsets face all of the same monitoring and measurement issues as voluntary registries described above. But in addition, once a carbon credit carries a market value and is legally equivalent to documented emissions reductions, two further issues rise to the fore—*additionality* and *permanence*.

Additionality refers to the certainty that a forest offset results in new carbon fixation, rather than simply subsidizing business as usual. Demonstrating additionality requires:

- A *baseline* against which new carbon stores can be measured. A projection of what would occur *over time* in the absence of project activities is the only acceptable baseline. Using a single pre-project quantity as a baseline might reward offset providers for sequestration that would have occurred in any case. Natural regeneration of abandoned farmland, for instance, could be used to offset continued fossil-fuel emissions, undercutting greenhouse gas reduction goals.
- Accounting for *leakage*, sometimes referred to as *secondary effects* or *displacement*. Leakage occurs when a project indirectly causes increased emissions outside the defined boundaries of the project itself. If an offset buyer pays to preserve forestland that is in imminent danger of paving over, for instance, but the development merely moves to a neighboring parcel, no net sequestration results. When exact measurements are impractical, leakage is often addressed by discounting, requiring that an offset seller fix more carbon than the quantity purchased in order to compensate for likely losses elsewhere.

Permanence is an issue because reduced emissions from a power plant or vehicle are by definition permanent. If fossil fuel remains unburned, the carbon it contains will never find its way into the atmosphere. If a sequestration project is to be considered fully equivalent to emissions reduction, it must fix carbon just as permanently. For forest offsets, permanence is complicated by the dynamic nature of ecosystems. Carbon stores ebb and flow during forest succession and with normal disturbance regimes, sometimes unpredictably in the case of fire, insect outbreak, or windthrow. However, permanence may be addressed through one of several mechanisms:

- Permanent easements on the land may impose a "lien" obligating the owner to maintain a guaranteed level of carbon stores indefinitely or for a contracted period of time.

- Offsets may be subject to a standard discount based on the risk of catastrophic carbon release.
- Offset contracts may be designed as short-term "leases," with payments made only so long as the carbon remains in place. When the contract expires, the buyer would need to replace this offset with an equivalent one.

In the absence of regulated markets, voluntary carbon trades are already occurring, with at least a dozen entities offering carbon offset services for a fee. Organizations are reducing or offsetting their "carbon footprint," and conferences are offering to offset attendees' air travel. The quality of such unregulated trades varies widely. It is tempting to see these voluntary trading systems as harmless, but they could establish misleading precedents for how a market might operate.

Other Policy Tools

It remains to be seen whether the issues with cap-and-trade systems can be resolved at a reasonable cost, allowing forest-based offsets to become tradeable commodities. In light of these uncertainties, we must also explore alternative policy options for increasing forest carbon stores. One approach to supplying public goods is for government agencies to produce them directly. For example, our national forests and other public lands might add carbon storage to the set of multiple uses they provide as a public service to the nation, through practices that accumulate carbon in old-growth forests, large woody debris, and forest soils.[4]

With 63 percent of our nation's forests privately owned, however, carbon-friendly management of public forestland will not be enough. A second policy approach would be for federal or state agencies to encourage private landowners to maintain or increase carbon stores through conservation payments channeled through the Wildlife Habitat Incentive Program (WHIP), Conservation Reserve Program (CRP), or Environmental Quality Incentives Program (EQIP). Such payments would help counter the tremendous financial incentives that favor forestry practices such as short rotations, high grading, and liquidation harvests, all of which yield maximum present value for timber while damaging long-term forest productivity and depleting carbon stores.

A third policy option is a sort of hybrid between a market and a public subsidy. Along with carbon markets, markets for wetlands, habitat, and water quality are emerging across the United States. Through these mechanisms, private restoration activities help mitigate damage from development activities. In the face of high transaction costs and low trading volume, some states use "in lieu fee" programs as an alternative to market trading, and these programs might offer viable models for forest carbon. In these programs, a state agency collects fees from those who damage wetlands, critical habitat, or water quality and uses the funds to finance restoration by private contractors, often accepting competitive bids. Similarly, a "no-net-loss" forest carbon policy could impose taxes or penalties on those who emit fossil-fuel carbon or release existing forest carbon stores, and use the revenue to subsidize increased forest carbon storage elsewhere. Already, Oregon requires new utilities to offset a portion of their carbon emissions, and many are purchasing offsets from The Climate Trust, a public-private entity that takes competitive bids from offset providers. Vermont's energy efficiency utility,

4 The carbon cycle of naturally fire-prone forests needs more investigation. Forests that naturally burn frequently might accumulate less carbon in the understory and on the forest floor, but more in large fire-resistant trees and long-lived charcoal.

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which offers assistance with efficiency investments financed through surcharges on utility bills, offers a similar model for a public-private solution.

Forest Carbon Controversies

Before we launch into either trading of forest carbon offsets or subsidies to boost forest carbon, we should be certain that the measures we pay for deliver the promised reductions in greenhouse gases. The questions discussed below concern three strategies that are often proposed as forest-based global climate change solutions: managing for fast-growing young forests, increasing carbon stored in wood products, and increasing use of woody biomass fuels. Any of these strategies, if employed without considering their full carbon-cycle impacts, could actually reduce carbon stores instead of increasing them.

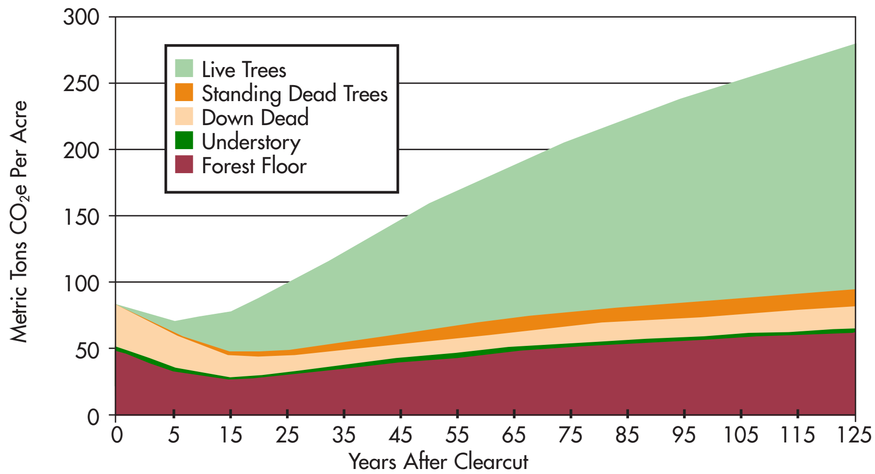
1: Does replacement of old, slow-growing forests with young, intensively managed plantations speed carbon sequestration?

Old forests represent large carbon sinks that need to be maintained as part of our nation's common infrastructure, much as we maintain our highways or our wetlands. Figure 7 shows the dynamics of carbon stores in a northeastern spruce-fir forest after an initial clearcut: an undisturbed forest continues to build new carbon stores well past a stand age of 125 years (the end point for this model though far short of the time required to create the complex structural conditions of old growth). Even though the rate of carbon sequestration may be faster in younger stands (the slope of the total carbon curve is steepest between 25 and 35 years post-clearcut), older forests do continue to add substantial carbon stores each year (the total carbon line is still rising rapidly at 125 years) and total carbon stored in the forest will be much higher with extended rotation ages. Under true old-growth conditions, windthrow and other natural disturbances will create patches of younger trees, but more carbon will likely be present in dead and downed material than would be found after commercial harvest. Additional research is needed to help us better understand carbon cycles under different forest types and management regimes.

Moving beyond abstract models to practices on the ground, harvesting methods clearly matter. Single-tree or small-group selection—which removes slow-growing trees, releases well-established but suppressed potentially vigorous trees, avoids soil damage, and leaves a high volume of standing trees—may in fact increase both live and dead carbon stores within a few years post-harvest. Conversely, a heavy cut that promotes regeneration-suppressing brambles or ferns, or a harvest that releases soil and litter carbon through erosion or accelerates respiration due to intense exposure, will likely suppress carbon fixation for several years or even decades. For the forest modeled in Figure 7, forest floor carbon declines for 15 years and down dead carbon for 45 years after a clearcut; regrowth of live trees and replacement of standing dead trees is also slow in early decades. Total carbon

FIGURE 7.

Non-Soil Forest Carbon, Northeast Spruce-Fir Stand



Data from Smith et al. 2006.

present in all five pools actually drops below the severely depleted levels present after a clearcut (year 0) for more than 20 years after the harvest.

Conversion of natural forests to intensively managed plantations may likewise release soil carbon as a byproduct of cultivation, burning, and soil drainage, and fertilizers that get new crops of seedlings off to a rapid start may release nitrogen oxides that are greenhouse gases several times more potent than CO_2.

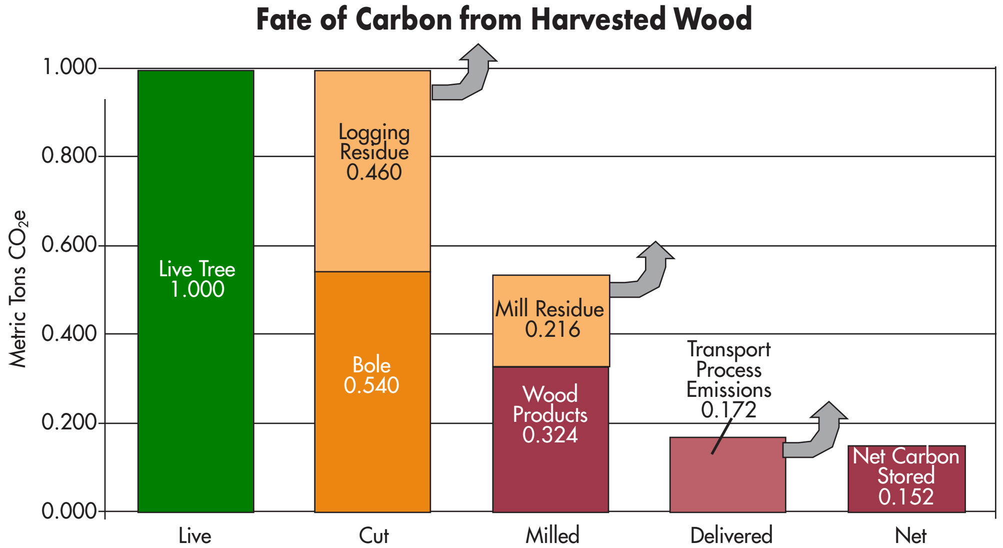
As Figure 5 illustrates, it is important to measure carbon system-wide, and not just in the forest itself. There would be no advantage to rapid carbon uptake by a young plantation if that carbon were quickly released once the trees were cut. Essentially each harvest shifts carbon from in-forest pools ("live vegetation" and "woody debris" pools in Figure 5—which continue to fix more carbon over time, though at a declining rate) to off-forest pools ("wood products" and "landfill" pools—which see slow, steady losses). To assess which strategy is more effective, it is important to track the whole system over time, including soil and dead biomass carbon in the forest and wood products outside the forest, which brings us to a second forest carbon controversy.

2: Does converting living trees into long-lived wood products increase carbon stores and reduce emissions?

Many forestland owners would like to operate their forests as carbon-fixation assembly lines, allowing trees to convert atmospheric carbon to wood, removing the live-tree carbon and storing it off-site, and releasing other trees from competition so that their growth and carbon storage rates increase. At face value, this claim seems convincing. However, a number of complexities underlie this simple explanation.

First, not all harvested carbon makes it into a finished wood product (Figure 8). Assume that a live tree containing 1 metric ton of CO_2e is cut (such a tree would contain about 0.27 metric tons of pure carbon or about 0.54 metric tons of dry material total). About 0.54 metric tons of CO_2e are in the bole, the portion transported to the mill (the exact proportion varies widely by region, forest type, and even market, and is generally lower in the Northeast). The remaining 0.46 metric tons CO_2e (the "harvest residue" flux in Figure 5 above) are left to rot and will do so fairly rapidly because they are stored in the smaller branches, leaves, and severed roots that now lie resting on or just under the forest floor. After passing through the primary mill and secondary processing facilities, ultimately about 60 percent of the bole, or 0.324 metric tons CO_2e, will be transformed into wood products. Like the logging slash left in the woods, the 0.216 metric tons of CO_2e in the slabs and sawdust will degrade fairly rapidly, likely either burned for fuel at the mill ("consumption" flux shown in Figure 5) or sold as garden mulch or animal bedding (part of the "wood products" pool in

FIGURE 8.



Data from Smith et al. 2006 and Gower et al. 2006.

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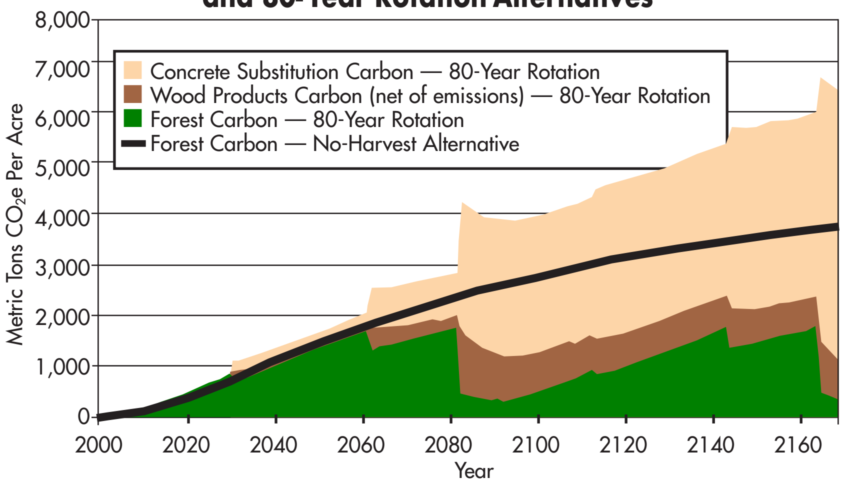
FIGURE 9.

Forest Ecosystem and Wood Products Carbon Under No-Harvest and 80-Year Rotation Alternatives



Adapted from Wilson 2006, data from Perez-Garcia et al. 2005.

Figure 5, but with a very short storage life). Emissions from both logging and mill residue take place over time, and the rate of release will vary with harvest methods, mill processes, and whether these parts of the tree decompose or are burned, but residence times in these pools are short relative to live trees or long-lived wood products.

Additional emissions of about 0.172 metric tons CO_2e result from harvest, transport, and processing,[5] mostly from burning of fossil fuels to run equipment, but also from less obvious sources like volatilization of finishes (the "processing" flux in Figure 5 should have an associated emissions flux to represent these costs of storing carbon in wood products). If burning of wood byproducts displaces fossil fuels in some processing and transport steps, as it does in many mills that use wood waste as an energy source, then this portion of emissions may be considered "carbon neutral" (see below, however, for some caveats). With losses at each step of the chain, the net gain in carbon stores may be little as 0.152 metric tons CO_2e—15.2 percent of the carbon originally stored in the live tree.

Depending on the type of wood product, carbon stores will continue to decay over time, with product half-lives ranging from 6 to 100 years (California Climate Action Registry 2007). If harvested wood products decay faster than standing or downed dead wood left in the forest (and the larger the tree, the slower the on-site decay), then harvesting wood is unlikely to increase carbon stores over time. Leaving trees to mature and die in place, making space and fertility for faster growth by their live neighbors, may in fact be a better carbon sequestration strategy.

Some of the most thorough research on wood products carbon has been conducted by the Consortium for Research on Renewable Industrial Materials (CORRIM), originally formed to analyze the life-cycle environmental impacts of wood compared to alternative building materials. Figure 9, developed by CORRIM researchers, provides one comparison of the "storage-on-the-stump" strategy with the "storage-in-wood-products" strategy. The figure shows projected carbon stores in a Pacific Northwest forest regenerated in the year 2000 under a no-harvest regime (black line) and an 80-year rotation with two thinnings (solid areas).

The no-harvest alternative (black line) clearly stores more carbon over time in the forest than the 80-year rotation. Under the harvested system, forest carbon (green area) fluctuates with standing timber volume, but never rises above 2,000 metric tons CO_2e per acre. Carbon in wood products (brown area) does accumulate over time, but slowly since many products decay by the end of each 80-year rotation.

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[5] Gower et al. (2006) found that nearly 1 ton of CO_2e is released for each ton of wood products produced. One ton of wood products contains about 0.5 tons of carbon, or 1.8332 tons CO_2e. So processing of wood emits about 53 percent as much CO_2e as is contained in the end products. Figure 8 reflects these losses, as processing results in emissions of 0.172 metric tons CO_2e in order to produce wood products that store 0.324 metric tons CO_2e.

The storage-in-wood-products strategy appears superior only if benefits include the substitution of wood for concrete in construction (tan area). Concrete manufacturing releases vast amounts of CO_2e, due to both fossil fuel used for heat and carbon released by the chemical transformation of lime to make cement. As Figure 9 illustrates, substituting wood for concrete would reduce CO_2e emissions dramatically; conversely, if management to boost forest carbon stores reduces the availability of wood for construction, it could inadvertently cause more emissions if builders turn to concrete or fossil-fuel-based plastics as substitutes.

However, adding concrete substitution benefits to forest and wood products stores on a single graph implies that one hundred percent of the wood harvested will displace concrete, a highly unlikely scenario since only 17.9 percent of new U.S. homes in 2005 used concrete in above-ground applications where wood substitution would be possible (Portland Cement Association 2006). A forest landowner who reports carbon sequestration benefits due to concrete substitution as part of a registry or who offers an offset sale that includes those benefits would need to prove that substitution actually takes place.

Once processing emissions and verified materials substitution are accounted for, credit for wood products carbon increases may be claimed by only one link in the chain—a chain that extends from the owner of the forestland where carbon was originally removed from the atmosphere, to the wholesaler, retailer, builder, and home-buyer, all of whom can claim they have reduced emissions by choosing wood over cement, steel, or other greenhouse-gas-emitting material. If increases in wood products carbon stores are to receive market payments or public subsidies, ownership of the credits will need to be clarified to avoid double counting.

3: Is woody biomass a carbon-neutral fuel?

Another wood product often promoted for its carbon sequestration benefits is woody biomass fuel. Many argue that woody biomass is by *definition* a carbon-neutral fuel because growing trees once fixed all the carbon that is eventually released by burning. The critical issue for carbon neutrality, though, is not past sequestration of carbon embodied in fuels, but whether releases are offset by *future* carbon stores. After all, fossil fuels too embody previously sequestered carbon in amounts equal to that released through burning. If climate change policy aims to moderate *future* concentrations of greenhouse gases, we should choose our renewable energy technologies for their *future* impacts.

Those who claim that woody biomass is *by definition* a carbon-neutral fuel make an unspoken assumption that the forest/generator system is maintained in a steady state. In a steady state, the amount of CO_2 released by harvesting and burning biomass would equal the amount fixed by the source forest over a period of time sufficient for the harvested trees to regrow. As always, however, the devil is in the details. How much fossil fuel is burned to harvest, chip, and transport the fuel? How severely and for how long is carbon fixation suppressed due to the impact of mechanized harvesting? How quickly do leaves, needles, and small branches left on-site rot and release their carbon stores? How quickly does residual vegetation respond with a spurt of rapid new growth?

Woody biomass can indeed be managed as a relatively carbon-neutral fuel. Just as wood may be a better option than concrete for use in building construction, substituting wood for fossil fuel use can be an important component of a national policy to mitigate climate change. In particular, emerging cellulosic ethanol technologies promise better ratios of energy output to input than conventional ethanol. But acceptance of tradeable carbon offsets based on substituting woody biomass for fossil fuels, or government subsidies for these fuels justified by their climate benefits, must require *continued* management of the source forest to fully replace the carbon removed, burned, and released. Once fixed, that carbon must remain stored (as living and dead forest material or as long-lived wood products) or must continue to offset fossil fuels in energy production. Furthermore, processing emissions must be accounted for. At some point in the future, as fossil fuels cease to be the norm for generating electricity, the business as usual baseline will change and there will be no further justification for trading offsets or offering subsidies for woody biomass.

Aside from complete and long-term accounting, standards for defining carbon neutrality of woody biomass fuel should incorporate common sustainable forestry practices to avoid unintended negative consequences. Vigorous biomass chip markets could provide perverse incentives to manage for the lowest common denominator in wood value. Operators bent on speedy processing of massive volumes of generic biomass are unlikely to use careful crop tree selection or directional felling to avoid residual stand damage. The Forest Stewardship Council and similar third-party certification systems already favor protection of a full suite of forest values, and it would be relatively straightforward to add carbon-neutrality of fuels derived from forests to their standards.

ATTACHMENT 1

Win-Win Forest Carbon Strategies: Restoration, Preservation, Sustainable Management

Given the difficulties with some proposals for boosting forest carbon, it seems prudent to support approaches that have few environmental drawbacks and many collateral benefits. Preventing forest conversion, replanting or restoring cleared or degraded forests, and lengthening rotations enjoy support from a wide variety of stakeholders, as these strategies also protect biodiversity, open space, water quality, remote recreation, and other increasingly threatened public values.

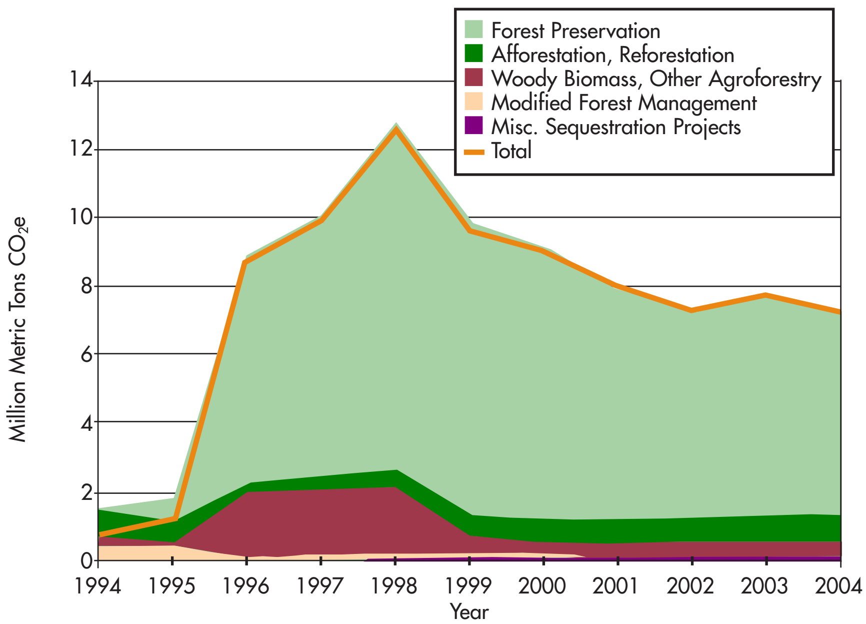
Forest preservation accounts for the great majority of carbon sequestration reported in DOE's 1605(b) registry, as Figure 10 shows. Registry guidelines permit preservation projects to claim 1/100th of the total CO_2e present in all carbon pools at the time of easement or fee purchase, plus report incremental carbon gains each year thereafter, so large quantities of sequestered carbon are registered immediately on project completion. Project sponsors must provide a permanent guarantee of forest cover through easements or other mechanisms, but are not required to prove that these lands would have been converted to other uses as strict additionality would require, so the CO_2 reductions attributed to forest preservation likely far exceed actual emissions reductions compared to a status quo baseline. However, where land conversion trends are well documented, this type of project provides tremendous potential for preventing carbon release due to forest losses.

Restoration—carbon sequestration through tree planting or regeneration (often called afforestation if land is naturally treeless or reforestation if temporarily cleared)—is the most easily documented means of boosting forest carbon stores, and the most commonly traded in the voluntary offsets marketplace. Eighty-three percent of the sequestration projects reported under the U.S. Department of Energy's 1605(b) program in 2004 involved tree planting (U.S. Department of Energy 2006). Figure 10 shows CO_2e sequestration reported to this registry in 2004; since reforestation project sponsors report the CO_2 sequestered in the reporting year, and tree-planting projects fix very little carbon in the early years, the large number of reforestation projects is not fully reflected in Figure 10.

Many reforestation projects are sponsored by electric utilities, which view forest offsets as a viable low-cost strategy to cope with coming climate change regulation. For example, two large-scale riparian forest restoration efforts sponsored by electric utilities have replanted bottomland hardwoods in the lower Mississippi River Valley. UtiliTree Carbon Company, founded by Edison Electric Institute and 41 utilities in 1995, has replanted 1,000 acres so far (some overseas) with a goal of sequestering 3 million metric tons of CO_2e. PowerTree

FIGURE 10.

Sequestration Projects Reported to U.S. Department of Energy



Data from U.S. Department of Energy 2006.

Carbon Company, formed by 25 power companies and several NGO partners in 2003, has spent $3.4 million to replant 3,600 acres and fix 2 million metric tons of CO_2e. Many of the "retail" carbon sequestration opportunities offered to individuals who want to offset personal carbon emissions also fund tree-planting programs. In the absence of national regulations, the quality of these programs varies tremendously. Valid reforestation offsets must include long-term verification that trees are alive and continue to grow.

Carbon sequestered through changes in forest management is perhaps the most difficult form of forest carbon enhancement to document, but it also holds great promise for secondary benefits to wildlife, water, and recreation. According to the North East State Foresters Association (2002), "management strategies that encourage larger trees, employ harvest methods that reduce waste and damage to residual trees, and minimize soil disturbance during harvest all improve carbon sequestration activities." The Pacific Forest Trust (Gordon 2006) estimates that "if managed over longer rotations [northeastern forests] can accumulate significantly more carbon, perhaps as much as 20 more tons (67 metric tons CO_2e) per acre. Neil Sampson (2004) estimates that improved forest practices such as longer rotations and higher stocking could increase CO_2e by 0.3 to 4.6 metric tons per acre per year in U.S. forests. Longer rotations could temporarily reduce wood supply and promote a shift to carbon-intensive substitutes, and this effect would need to be carefully monitored. But over time, harvest volume from such forests would recover and could even increase.

Potential for New Collaborations

As high fossil fuel use is the ultimate cause of human-induced global climate change, the ultimate solution depends upon reduced use of those fuels through energy efficiency and renewable substitutes. Given our addiction to oil, coal, and natural gas, however, that transition will be costly and time-consuming, and restoring forest carbon stores can help buy time. A national policy to enhance forest carbon stores offers an opportunity for collaboration among unusual allies—regional, national, and international environmental NGOs; small woodlot owners; the National Forest system; forest ecologists; and foresters, logging contractors, and the wood products industry. These groups have a shared interest in moderating climate change, protecting forestland from conversion, understanding the dynamics in natural forest systems, maintaining timber stocks in working forests, and promoting use of long-lived wood products.

Because of this congruence of diverse interests, forest carbon sequestration will likely be an important part of an emerging national climate change policy for the United States. Yet if forests are to make a significant and lasting contribution, and if we are to avoid unintended damage to other natural processes and values, it is critical for both accounting systems and policy measures to be designed with great care. We need improved carbon monitoring techniques, at both national inventory and project levels. Then we should begin to test and study forest sequestration with projects that provide broadly acknowledged secondary public benefits and few possible drawbacks. Overall, we need to keep forests as forests, restore them to a state of health, and manage them to maintain high volumes of above- and below-ground carbon. As an added bonus, these measures will help promote a more resilient forested ecosystem, better able to withstand the climate changes that have already begun.

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ATTACHMENT 1

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COVER PHOTOS:
Top: Hearts Content Scenic Area, Allegheny National Forest, Pennsylvania. Globally, forest losses account for nearly one-quarter of greenhouse gases released due to human activities. Efforts to reduce our climate impacts need to include protection for forest carbon stores like those in this rare eastern old-growth stand.
Photo by Donald L. Gibbon

Left: Fossil fuel combustion is the primary source of excessive greenhouse gases in the atmosphere, and coal-fired power plants are still on the increase.
Photo: Corbis Images

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ATTACHMENT 2

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Pyrogenic carbon emission from a large wildfire in Oregon, United States

John Campbell,[1] Dan Donato,[1] David Azuma,[2] and Beverly Law[1]

Received 21 March 2007; revised 1 September 2007; accepted 25 September 2007; published 27 December 2007.

[1] We used a ground-based approach to compute the pyrogenic carbon emissions from the Biscuit Fire, an exceptionally large wildfire, which in 2002 burned over 200,000 ha of mixed conifer forest in southwestern Oregon. A combination of federal inventory data and supplementary ground measurements afforded the estimation of preburn densities for 25 separate carbon pools at 180 independent locations in the burn area. Average combustion factors for each of these pools were then compiled from the postburn assessment of thousands of individual trees, shrubs, and parcels of surface and ground fuel. Combustion factors were highest for litter, duff, and foliage, lowest for live woody pools. Combustion factors also increased with burn severity as independently assessed from remote imagery, endorsing the use of such imagery in scaling emissions to fire area. We estimate the total pyrogenic carbon emissions from the Biscuit Fire to be between 3.5 and 4.4 Tg C (17 and 22 Mg C ha^{-1}) depending on uncertainty in our ability to estimate preburn litter pools and mineral soil combustion with a central estimate of 3.8 Tg C (19 Mg C ha^{-1}). We estimate that this flux is approximately 16 times the annual net ecosystem production of this landscape prior to the wildfire and may have reduced mean net biome production across the state of Oregon by nearly half in the year 2002.

Citation: Campbell, J., D. Donato, D. Azuma, and B. Law (2007), Pyrogenic carbon emission from a large wildfire in Oregon, United States, *J. Geophys. Res.*, *112*, G04014, doi:10.1029/2007JG000451.

1. Introduction

[2] Efforts to quantify carbon exchange between terrestrial vegetation and the atmosphere have typically focused on patterns of photosynthesis and respiration. While complex in nature, basic mechanistic understanding of physiology and soil processes has been used in models to predict vegetation responses over broad spatial and temporal domains. In contrast, pyrogenic releases of carbon from vegetation to the atmosphere, while physically simple, are inherently stochastic and therefore not typically included in most process-based models [*Schimel and Baker*, 2002; *Arora and Boer*, 2005].

[3] This deficiency in global vegetation modeling was made apparent following the El Niño of 1997–1998 when an anomalous two-fold increase in global atmospheric CO_2 enrichment was attributed to pyrogenic emissions from Southeast Asian wildfires [*Page et al.*, 2002; *van der Werf et al.*, 2004]. Interest in this phenomenon, combined with advances in remote detection of wildfire [*Lentile et al.*, 2006], concerns over fuel-driven increases in fire frequency and severity in the western United States [*Schoennagel et al.*, 2004], and possible feedbacks between global warming and wildfire frequency [*Westerling et al.*, 2006] has resulted in a number of large-scale, bottom-up efforts to quantify pyrogenic emissions from Africa [*Barbosa et al.*, 1999], Alaska [*French et al.*, 2002; *Kasischke and Bruhwiler*, 2002; *French et al.*, 2004], Siberia [*Soja et al.*, 2004], China [*Lü et al.*, 2006], and North America [*Wiedinmyer et al.*, 2006]. All of these studies use the same general measure-and-multiply approach popularized by *Seiler and Crutzen* [1980], where pyrogenic emissions are calculated as the product of four parameters: area burned, fuel density (biomass per unit area), combustion factor (fraction of biomass consumed by fire), and emission factor (mass of a given chemical species released per mass of fuel consumed). For the most part, the area affected by fire can be accurately assessed either remotely or from inventories and there is general agreement on the emission factors for carbon and other airborne pollutants. However, while most studies recognize the need to vary the inputs of fuel density by vegetation type and the combustion factors by fire severity, the ground data needed to parameterize these functions has been deeply lacking. This is especially true for combustion factors that are compiled from a limited source of widely varying data [see *Peterson and Sandberg*, 1988; *Soja et al.*, 2004; *Wiedinmyer et al.*, 2006] and simple assumptions on how these factors vary with respect to an operationally defined fire severity classification. To improve our regional and global estimates of pyrogenic emissions, it is necessary to improve the specificity and accuracy of our

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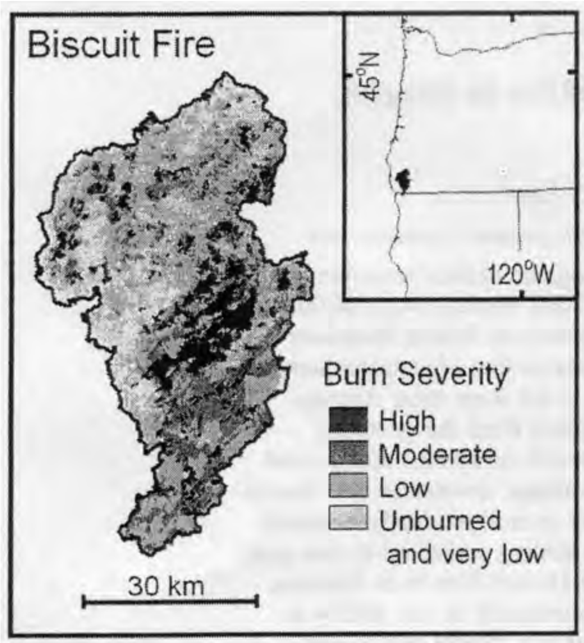

Figure 1. The Biscuit Fire. The Biscuit Fire burned at a mix of severities over 200,000 ha of forest in the Siskiyou Mountains of southwestern Oregon and northern California in the summer of 2002 making it the largest contiguous wildfire in Oregon history. The severity classes shown are those of the remotely derived 2002 BAER classification.

estimates of fuel density and combustion factors beyond what is generally available [*Houghton et al.*, 2000], especially for temperate ecosystems where quantification of fire effects lags behind that of boreal systems.

[4] In this study we consider an exceptionally large wildfire, the Biscuit Fire, which in 2002 burned over 200,000 ha of mixed conifer forest in southwestern Oregon. Carbon emissions from a fire this large are likely to contribute sizably to the annual carbon budget of the region [*Law et al.*, 2004]. Accurate quantification of this flux has been limited by our understanding of the amount of fuel present and the fraction actually combusted. Conveniently, however, the Biscuit perimeter encompassed 180 systematically located U.S. Department of Agriculture (USDA) Forest Service inventory plots. Structural measurements made on these plots before and after the fire, combined with biomass measurements on additional plots, now afford an assessment of preburn fuel density and combustion factors across a combination of forest types, ages, and burn severities unprecedented for a single fire.

[5] Our objectives were to: (1) Determine combustion factors (as a probability distribution) for each of 25 different forest carbon pools representing different fuel types. (2) Assess variation in the above combustion factors as a function of remotely sensed burn severity. (3) Combine the combustion factors with estimates of preburn fuel densities and burn area by severity to estimate fire-wide pyrogenic carbon emission. (4) Assess the utility of federal inventory plots as a method of compiling much needed fuel density and combustion factors. Results are then considered in the context of regional carbon fluxes over time for the same forest and throughout the region in the year of the fire.

2. Methods

2.1. Study Sites

[6] The Biscuit Fire burned at a mix of severities across 200,000 ha of forest in the Siskiyou Mountains of southwestern Oregon and northern California in the summer of 2002, making it the largest contiguous wildfire on record for Oregon (see Figures 1 and 2). The Siskiyou Mountains are characterized by a variety of forest types from Douglas-fir/western hemlock/bigleaf maple communities on mesic sites, to Douglas-fir/tanoak on drier sites, to Jeffrey pine on ultramafic substrates [see *Whittaker*, 1960].

[7] Within the perimeter of the Biscuit Fire there are 180 regularly spaced permanent federal inventory plots (i.e., systematic sample design). In these one-hectare plots (referred to hence forth as inventory plots), metrics to quantify biomass, composition, and various structural attributes have been collected in approximate 10-year intervals since 1970 [see *USDA*, 1995]. The most recent measurements before the Biscuit Fire were made between 1993 and 1997. A 2003-2004 measurement cycle in the years following the fire was then conducted in which additional metrics quantifying fire effects were collected [see *USDA*, 2003].

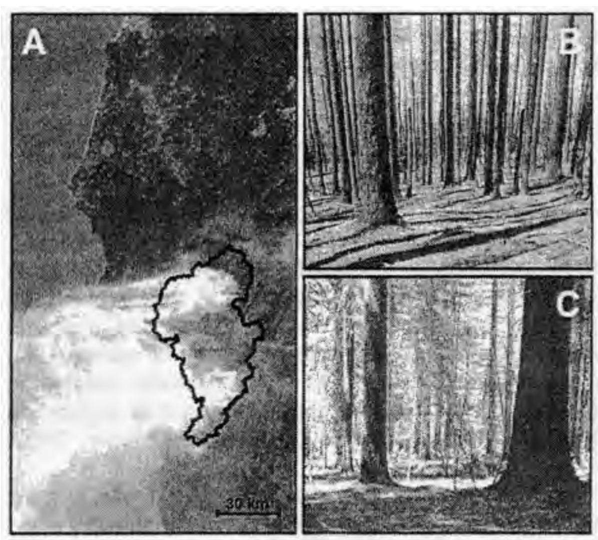

Figure 2. Images from the Biscuit Fire showing (a) the smoke plume drifting over the Pacific Ocean, (b) a forest stand which burned at high severity, and (c) a forest stand which burned at low severity. The black line on Figure 2a denotes the final perimeter of the fire. Even in the most severely burned stands in the Biscuit, where mortality reached 100% and fine surface fuels were completely combusted, tree boles and fine branches remained largely intact. Typical low severity burn in the Biscuit was characterized by bole scorching, minimal canopy mortality, and partial consumption of understory vegetation and ground fuels. Photo for Figure 2a provided by NASA Visible Earth (http://visibleearth.nasa.gov/); photos for Figures 2b and 2c courtesy of Joe Fontaine and Dan Donato

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[8] While data from the inventory plots provided detailed measurements of fire effects on the boles and crowns of most trees, as well as most detritus pools, they did not include fire effects on coarse woody detritus and smaller woody stems killed in the fire. To assess the effects of the Biscuit Fire on these carbon pools, we made pertinent measurements (see below) in 2004 on 54 additional one-hectare plots (referred to hence forth as supplementary plots) randomly located within 54 independent forest stands deliberately distributed across burn severities, including areas unaffected by fire.

2.2. Pyrogenic Emissions

[9] Following the approach of *Seiler and Crutzen* [1980], pyrogenic carbon emissions from the Biscuit Fire were computed according to equation (1):

$$PE = \sum_{i=1,j=1}^{n} A_i (D_{ij} \cdot CF_{ij}) \qquad (1)$$

where *PE* is pyrogenic emission in mass of carbon, *A* is the area affected by burn severity class *i*, *D* is the preburn carbon density in mass per unit area of carbon pool *j* averaged across plots of burn severity *i*, and *CF* (hence forth referred to as combustion factor) is the fraction of preburn carbon pool *j* combusted in burn severity class *i*. In this study we recognize four burn severities: high, moderate, low, and unburned/very low; and 25 separate carbon pools separated by tissue type, growth form, size class, and mortality status.

2.3. Pool-Specific Combustion Factors

[10] The methods for calculating combustion factors specific to various carbon pools are shown in Table 1. We used two basic approaches for arriving upon combustion factors: (1) a back-calculation method where combustion factors are calculated solely from postburn measurements of charring and perceived loss of foliage and branches, and (2) a before-and-after method where combustion factors are calculated as the difference between preburn and postburn mass. As a general rule, the combustion factor of large carbon pools and those that experience low fractional combustion (i.e., live stem wood) are more precisely assessed using the back-calculation method since the sampling error associated with before-and-after comparisons would result in unacceptably low signal-to-noise ratios. Conversely, the combustion factor of smaller carbon pools and those that experience high fractional combustion (i.e., fine woody debris and surface litter) are more precisely assessed using the before-and-after method since postburn measurements reveal little regarding the preburn pool size.

[11] For each separate carbon pool, combustion factors were assessed at the finest possible scale (see Table 1). For instance, since the impacts of fire on foliage, bark, and stem wood were measured separately on each tree, combustion factors for these pools were computed separately for each tree. When measurements represented plot-level average responses (e.g., downed wood), combustion factors were computed at the plot level.

[12] Unlike tissue combustion in larger trees, much of the losses in smaller trees (<7 cm DBH; diameter at 1.37 m above ground) occurs as a result of complete tree combustion. To quantify the incidence of complete combustion of small diameter trees, the frequency of small conifers was compared between burned and unburned plots. The apparent deficit of small diameter trees in burned plots was attributed to complete combustion (see Table 1). Similarly, we investigated the need to account for complete combustion of stumps and other coarse woody detritus, which was not assessed in the postburn inventory. However, despite anecdotal evidence of complete combustion of stumps and logs, there was no detectable difference in these pools between burned and unburned plots; consequently carbon losses due to their complete combustion are believed to be trivial.

2.4. Preburn Carbon Density

[13] Preburn carbon density for each recognized carbon pool was computed for each inventory plot using preburn survey data and a combination of allometric scaling equations appropriate for species in the region. Tree bole mass was estimated with species- and site-specific allometric equations relating stem diameter to volume and species-specific wood density values [*van Tuyl et al.*, 2005]. Foliage and bark mass were estimated directly from species- and site-specific allometric equations [*Means et al.*, 1994]. The mass of downed woody detritus was computed from line intercept data using geometric scaling and species-specific wood density values [*Harmon and Sexton*, 1996]. Biomass of small hardwoods (including shrubs) was determined using allometric equations derived empirically from tissue harvests made in the region of the Biscuit Fire: stem mass in g = 2203(1 − exp(−0.0002(shrub volume in dm^3))); foliage mass in g = 6498(1 − exp(−0.0001(shrub volume in dm^3))). Ocular estimates of total grass and forb coverage was converted to biomass using 4.0 g m^{-2}, which is the average mass per unit coverage reported for common local species [*Means et al.*, 1994].

[14] Because litter and duff masses were not recorded on the inventory plots prior to the fire, it was necessary to estimate preburn masses for these pools from samples collected in 2004 from locations distributed throughout the Biscuit area but unaffected by fire. Recognizing that these preburn carbon pools varied across the forests affected by the Biscuit, we originally set out to collect unburned litter and duff samples from a variety of cover types and apply these cover type-specific masses to each inventory plot according to the plot's location on a cover type map. However, upon collecting these samples it became apparent that both inaccuracies in the cover type map and variability in forest floor (soil O-horizon) depth within forest type were leading to false accuracy. Considering this, we decided to aggregate forest types on the Biscuit into the two most distinct classes: (1) low biomass forests growing on ultramafic (serpentine) substrates, and (2) higher biomass forests growing on nonultramafic substrates. Sampling involved the collection of six-inch-diameter parcels of forest floor from 8 to 32 points from each of 43 independent plots distributed throughout the Biscuit perimeter (11 in ultramafic sites, 32 in nonultramafic sites). Samples were dried, separated into duff and litter, and produced four separate values: 1691 and 993 g m^{-2} for litter and duff on ultramafic substrates, respectively; 2000 and 1399 g m^{-2} for litter and

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Table 1. Methods and Decision Rules for Computing Combustion Factors for Various Carbon Pools'

Carbon Pool	Method for Deriving Combustion Factor	Sample Size and Source
Foliage (large live trees)	The fraction of foliage reported missing from each tree via ocular estimate was equated to the fraction combusted and then corrected to account for foliage killed and dropped but not combusted based on postbum measurements of new litter accumulation.	13,000 trees in inventory plots
Branch (large live trees)	The fraction of branch and twigs reported missing from each tree via ocular estimate in the inventory records was equated to the fraction combusted.	13,000 trees in inventory plots
Bark (large live trees)	Computed for each tree as the product of: fraction of bole surface charred (derived from fire scar measurements), fraction of bark depth charred (determined through supplementary measurements to average 0.29 independent of fire severity), and fraction of mass loss resulting from charring (assumed to be 0.9, 0.5, 0.4 for high, moderate, and low severity plots, respectively crudely extrapolated from *Czimcalk et al.* [2002] and assuming a maximum bark temperature of 500°C).	13,000 trees in inventory plots
Bole (large live trees)	No bole wood consumption was reported in either the inventory or supplementary plots for these larger live trees. Therefore, combustion was assumed to be negligible.	not applicable
Bole, bark, branch, and foliage (small live conifers)	Based on a comparison of density and size class distribution between burned and unburned plots, complete combustion of all tissues was determined to occur at a frequency of 0.6, 0.6, and 0.4 for high, moderate, and low severity plots, respectively. Bark, branch, and foliage loss for trees not fully combusted was assumed to be equal to that of larger trees.	430 trees in supplementary plots
Bole, bark, branch, and foliage (small live hardwoods)	Tissue combustion was determined for each stem as the difference between postburn volume (computed allometricly from basal diameter and stem height) and preburn volume (extrapolated allometricly from postburn basal diameter).	480 trees in supplementary plots
Bole, bark, branch, (standing dead trees)	Tissue combustion was computed by the same methods used for live trees except that in cases where bark was absent surface char was assessed as wood rather than bark combustion. Field records of char depth, while variable, indicate no difference between live and dead trees.	1,200 trees in inventory plots
Downed dead wood (large)	A lack of data on char severity for large downed wood prevented direct assessment. Instead the combustion factors for large downed wood was assumed to be twice that of standing dead wood.	not applicable
Downed dead wood (medium and small)	Fraction combusted was determined for each plot as the difference between prebum and postbum debris volume (determined line intercept transects).	180 inventory plots
Litter (0;-horizon, including leaf litter and woody fragments <0.51 cm diameter)	Computed occular estimates of bum effects on I3.5m2 plots as $(a + 0.5b)/c$ where, a is the sum area of all sublitter surfaces indicating total litter combustion (light and deeply charred duff, mineral soil and rock), b is the area over which litter was reported as lightly charred, and c is total area believed to be covered by litter prior to the fire (the sum of all surfaces covered by uncharred litter, lightly charred litter, and all sublitter surfaces showing some charring).	720 inventory subplots
Duff (O_e and O_a - horizon)	Computed from postburn surveys with the same equation used for litter substituting duff char values for that of litter and referring only to subduff layers as indicators of duff loss.	720 inventory subplots
Mineral soil (A and B - horizon including fine roots to 10 cm)	Combustion of mineral soil C was assessed only when postbum surveys reported either a deeply charred mineral surface (in which case all C in the top 4 cm of soil was presumed combusted) or a moderately charred mineral surface (in which case all C in the top 2 cm of soil was presumed combusted).	720 inventory subplots

'Large refers to >7.62 cm DBH for trees and fragment diameter for dead wood; Small refers to <7.62 cm DBH for trees and fragment diameter for dead wood. Sample size refers to the number of independent events assessed across the fire. For details regarding postfire sampling procedures, see *USDA* [2003].

duff on nonultramafic substrates, respectively. *To* verify our estimates *of* preburn litter and duff were reasonable, we compared our numbers to modeled estimates using the FCCS national fuel bed map and associated fuel loadings *[Sandberg et al.,* 2001; *Ottmar et al., 2007]* (http://www.fs.fed.us/pnw/feralfccs). *As* shown in Table *2, differ*ences in cover type partitioning between that *of* our study and that *of* the FCCS do not permit comparisons at scales smaller then the entire fire. When comparing values across the entire Biscuit, our values for duff mass were lower than that *of* FCCS and our values for litter mass were higher than that of FCCS suggesting a discrepancy in the operational definition of litter and duff between the two tnetbodologies. However, the sum of litter and duff (i.e., forest floor) is in general agreement between the two approaches with the FCCS predicting only 30% more mass fire wide than we estimated from our sampling.

[Is] *A* considerable portion of the Biscuit reburned the 38,000-hectare 1987 Silver Fire, introducing the possibility that fuel masses were different for these parts of the Biscuit. However, the pre-Biscuit inventory was conducted between 1993 and 1997, 6—11 a after the Silver Fire; thus most such differences were implicitly accounted for line the inventory plot data. *As* for litter and duff masses, which were not

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Table 2. A Comparison of Modeled Forest Floor Mass to That Measured for This Study

Forest Cover Type	Fraction of Biscuit Area	Preburn C Pool, kg C ha^{-1}		
		Litter	Duff	Total Forest Floor[a]
Modeled from FCCS database[b]				
(2) W.hemlock/W.redcedar/Douglas-fir	0.53	4000	21075	25075
(7) Douglas-fir/Sugar pine/Tanoak	0.15	1277	21523	22800
(28) Ponderosa pine savanna	0.09	986	4078	5064
(38) Douglas-fir/Madrone/Tanoak	0.09	3193	8291	11484
(10,24,47,48,52,53,59) All others	0.14	2426	38596	41022
All combined and weighted by class	1.00	2989	19663	22652
From field measurements in this study				
Forest on nonultramafic substrates	0.72	10001	6993	16994
Forest on ultramafic substrates	0.28	8455	4966	13421
All combined and weighted by class	1.00	9562	6417	15979

[a]The sum of litter amd duff.
[b]Number codes correspond to mapped FCCS fuel bed types.

measured in the pre-Biscuit inventory and were derived from our supplementary sampling, the absence of unreburned Silver Fire area prohibited direct sampling of this condition to assess forest floor masses in those stands prior to reburning. We addressed this issue by collecting forest floor samples from the nearby Galice Fire, which burned the same year as the Silver Fire but did not reburn in the Biscuit. Litter and duff masses in the Galice were not discernibly different from those collected from unburned sites, suggesting that the forest floor in the Silver area had recovered to preburn levels by the time the Biscuit burned.

[16] An estimate of the carbon present in the top 10 cm of mineral soil throughout the area affected by the Biscuit was based on a rock-free soil carbon fraction of 0.10, a rock-free soil bulk density of 0.89 g cm^{-3}, a fine root mass of 0.01 g cm^{-3}, (determined from 96 soil cores taken on 3 unburned plots) and a rock fraction of 0.50 by volume chosen to represent both the typical and highly skeletal substrates present in the Siskiyou mountains. We assumed the carbon content of all pools to be 0.50 by mass (a standard approximation) except for the litter and duff pools which we assumed to be 0.40 (based on Dumas combustion of 36 field samples producing an average of 0.40 and a standard deviation of 0.08).

2.5. Binning of Data by Burn Severity

[17] To assess carbon combustion as a function of burn severity, each of the study plots was classified as one of four burn severities (e.g., high, moderate, low, or unburned/very low) based on an overlay of the Biscuit BAER (Burned Area Emergency Rehabilitation) fire severity map. The levels of severity in the BAER map were based on classification of the differenced normalized burn ratio (dNBR), a widely used index of burn severity derived from Landsat data [*Miller and Yool*, 2002; *van Wagtendonk et al.*, 2004; *Key and Benson*, 2005]. dNBR is a measure of prefire to postfire change in the ratio of near- to short-wave infrared spectral reflectance [*Key and Benson*, 2005]. BAER assessments are used by federal land management agencies for remediation reconnaissance and are independent of any of the measurements used to compute combustion in this study. Then each of the approximately 60,000 separate combustion computations made for individual trees, plots of ground cover, or debris transects were binned by the burn severity of the plot in which the record was taken and averaged to produce the values CF_{ij} in equation (1). This approach allowed us to assess the ability of BAER severity classification to detect within-fire variability in the combustion of various carbon pools and therefore the utility of BAER severity in scaling combustion factors for other fires. Similarly, to account for possible interaction between preburn carbon density and subsequent burn severity, the preburn carbon densities of each for each plot were averaged by BAER severity classification to produce the values Dij in equation (1). Finally, the total area affected by each burn severity class in the Biscuit Fire perimeter (value A_i in equation (1)) was determined from the BAER severity map to be 32, 46, 84, and 41 thousand ha for the high, moderate, low, and unburned/very low severities, respectively. While several different burn severity maps are available for the Biscuit, we chose BAER because it is among the most readily available and widely used burn severity classification for wildfires in the western United States.

3. Results

3.1. Combustion Factors

[18] The combustion factors estimated for each carbon pool and burn severity class are shown in Table 3. Discrepancies between mean and median values indicate a right skew in the event probability in high severity plots and a left skew in the lower severity plots. In other words, while combustion scales to the landscape according to the average of that experienced by individual trees or specified patches of litter, most individuals in low severity plots are affected by fire to a much lesser degree than the average of individuals located in low severity plots. Conversely, most individuals in high severity plots are affected by fire to a much greater degree than the average of individuals located in high severity.

[19] Nearly all 25 carbon pools show a monotonic increase in combustion factor as burn severity increases from the unburned-very low class through to the high severity class (Table 3). Such a consistent trend for ground, surface, and canopy fuels is an endorsement of the BAER severity classification for distinguishing the fraction of carbon combusted from different pools. Such trends are especially clear in the highly combustible ground and surface pools such as litter and fine woody detritus. This relationship between remotely assessed fire severity and ground and

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Table 3. Average (and Median) Combustion Factors by Carbon Pool and Burn Severity

Forest Carbon Pool (Fuel Type)	Combustion Factor[a] High Severity		Moderate Severity		Low Severity		Unburned and Very-Low Severity	
Foliage								
Large conifers	0.69	(0.98)	0.27	(0.01)	0.08	(0.00)	0.02	(0.00)
Large hardwoods	0.58	(0.87)	0.29	(0.00)	0.12	(0.00)	0.03	(0.00)
Small conifers	0.89	(1.00)	0.76	(1.00)	0.44	(0.07)	0.01	(0.00)
Small hardwoods	1.00	(1.00)	0.80	(1.00)	0.50	(0.00)	0.00	(0.00)
Grass and forbs	1.00	(1.00)	0.76	(0.88)	0.75	(0.87)	0.70	(0.83)
Branch								
Large conifers	0.05	(0.08)	0.02	(0.00)	0.00	(0.00)	0.00	(0.00)
Large hardwoods	0.05	(0.06)	0.02	(0.00)	0.01	(0.00)	0.00	(0.00)
Small conifers	0.64	(1.00)	0.69	(1.00)	0.41	(0.00)	0.00	(0.00)
Small hardwoods	0.79	(0.81)	0.63	(0.65)	0.40	(0.41)	0.00	(0.00)
Bark								
Large conifers	0.20	(0.26)	0.06	(0.03)	0.03	(0.01)	0.01	(0.01)
Large hardwoods	0.22	(0.26)	0.11	(0.15)	0.03	(0.00)	0.01	(0.00)
Small conifers	0.70	(1.00)	0.70	(1.00)	0.42	(0.05)	0.01	(0.01)
Small hardwoods	0.79	(0.81)	0.63	(0.65)	0.40	(0.41)	0.00	(0.00)
Bole								
Large conifers	0.00	(0.00)	0.00	(0.00)	0.00	(0.00)	0.00	(0.00)
Large hardwoods	0.00	(0.00)	0.00	(0.00)	0.00	(0.00)	0.00	(0.00)
Small conifers	0.61	(1.00)	0.68	(1.00)	0.40	(0.00)	0.00	(0.00)
Small hardwoods	0.79	(0.81)	0.63	(0.65)	0.40	(0.41)	0.00	(0.00)
Dead wood								
Large standing	0.12	(0.07)	0.04	(0.03)	0.02	(0.01)	0.02	(0.00)
Small standing	0.61	(1.00)	0.68	(1.00)	0.40	(0.00)	0.00	(0.00)
Large downed	0.24	(0.14)	0.08	(0.06)	0.04	(0.02)	0.04	(0.01)
Medium downed	0.79	(1.00)	0.73	(0.83)	0.67	(0.76)	0.62	(0.67)
Small downed	0.78	(0.83)	0.58	(0.62)	0.61	(0.70)	0.62	(0.69)
Forest floor and soil[b]								
Litter	1.00	(1.00)	0.76	(0.88)	0.75	(0.87)	0.70	(0.83)
Duff	0.99	(0.99)	0.51	(0.64)	0.54	(0.75)	0.44	(0.50)
Soil to 10 cm	0.08	(0.05)	0.04	(0.01)	0.04	(0.01)	0.02	(0.00)

[a]Fraction *of* prebum mass lost to combustion.
[b]Litter is O; horizon, duff is O_e and O_a horizon, soil is all mineral soil to a depth *of 10* cm including fine roots. For live trees, small is <7.62 cm DBH; large is >7.62 cm DBH. For dead wood, small is 0.51—2.54 cm, medium is 2.54—7.62 cm, and large is >7.62 cm diameter.

surface fuel combustion was not a foregone conclusion, as fire effects on the ground can often be decoupled from fire effects in the canopy *[Pyne et al., 1996; van Wagner, 1977].* While litter, duff, and small woody detritus combustion was lowest in the unburned-very low severity plots, the fact that the values still average *60%* combustion indicate just how prevalent surface fire was across all of. the Biscuit Fire. Field records confirm that, of the 41 inventory plots that were remotely classified as unburned-very low, only two showed no sign of surface fire.

[2o] Combustion factors also varied expectedly among carbon pools. Pools with larger surface to volume ratios (e.g., foliage, small stems, and litter) showed consistently higher combustion factors than those with lower surface to volume ratios (e.g., large tree boles). This is consistent with most fire behavior models which equate fuel fragment size inversely to their propensity for desiccation and combustibility *[Reinhardt et al., 1997].*

3.2. Preburn Carbon Pools

[211 Preburn carbon mass for each pool and burn severity class is shown in Table 4. As is the case with most mature forest landscapes, biomass is concentrated in the largest trees. Differences in biomass among burn severities reflect the tendency for stands with more small trees and fewer large trees to burn at higher severity, a finding consistent with that of *Azuma et al. [2004].* Notably, this trend is reversed for dead wood in that higher severity plots had consistently lower amounts of coarse woody detritus prior to the fire. To aid in comparison with other wildfire research [e.g., *Ottmar et al., 2007],* prebum carbon pools were also summarized according to conventional fuel categorization and expressed in total dry mass per unit area along with corresponding combustion factors in Table 5.

3.3. Total Pyrogenic Emissions and Sources

[22] Using equation (1) to combine the combustion factors of Table 3, the prebum carbon pools of Table *4,* and the area exposed to each burn severity class (see methods above) yields a Biscuit-wide pyrogenic emission of 3.8 Tg C. Here, the two largest sources of pyrogenic emissions were both from the forest floor. As shown in Table *6,* 31% of the total pyrogenic emissions arose from combustion of the litter layer and another *26%* arose from combustion of the underlying duff and mineral soil layers. The next largest source was the combustion of dead wood which contributed 19% to total emissions. The relative contribution of different pools to total emissions was largely the same when carbon losses were computed separately by burn severity class, with the litter and duff pools being the largest contributors. However, as burn severity decreases there is a slight shift in major combustion sources from the canopy

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Table 4. Average Carbon Density by Forest Carbon Pool and Burn Severity[a]

	Carbon Density, kg C ha^{-1}				
Forest Carbon Pool	High Severity	Moderate Severity	Low Severity	Unburned Very Low Severity	All Burn Severities
Foliage					
Large conifers	2853	3045	3397	3670	3242
Large hardwoods	1152	234	1594	3813	1698
Small conifers	1172	3272	1746	1260	1863
Small hardwoods	378	397	431	461	417
Grass and forbs	3	2	2	3	2
Branch					
Large conifers	11421	6725	9886	11399	9858
Large hardwoods	2759	565	3964	10113	4350
Small conifers	105	117	2152	64	609
Small hardwoods	505	432	831	549	579
Bark					
Large conifers	8759	7279	12171	16587	11199
Large hardwoods	2779	565	4053	10694	4523
Small conifers	99	89	2148	52	597
Small hardwoods	18	115	67	76	69
Bole					
Large conifers	40650	38509	65120	85396	57419
Large hardwoods	19331	3991	28727	70943	30748
Small conifers	347	365	236	202	288
Small hardwoods	188	1127	711	772	700
Dead wood					
Large standing	6791	2877	7338	6701	5927
Small standing	869	554	2148	2998	1642
Large downed	6179	9003	12145	7201	9324
Medium downed	1388	1422	1933	2196	1798
Small downed	1055	1414	1499	2028	1543
Forest floor and soil					
Litter	9228	9096	9743	9929	9499
Duff	5979	5806	6655	6898	6335
Soil and roots to 10 cm	45500	45500	45500	45500	45500

[a]Values are the average of 26, 41, 66, and 43 inventory plots for high, moderate, low, and unburned-very low severity study plots, respectively, except that one Biscuit-wide value was used for soil and roots. For live trees, small is <7.62 cm DBH; large is >7.62 cm DBH. For dead wood, small is 0.51–2.54 cm, medium is 2.54–7.62 cm, and large is >7.62 cm diameter. Litter is O_i horizon; duff is O_e and O_a horizon.

to the ground and surface, reflecting the shift in fire behavior from a crown fire (which in most cases included ground and surface combustion as well) to a surface fire.

3.4. Uncertainty Assessment

[23] The sources of uncertainty in our estimates of pyrogenic emissions range from measurement uncertainty in the field, to sampling error at both the plot and landscape level, to the various quantitative assumptions regarding allometric scaling of preburn carbon pools and mass losses, to decision rules regarding the partitioning of carbon pools. Considering the difficulty in estimating combustion of subsurface carbon and that 65% of the total fire-wide carbon emissions may come from the combustion of litter, duff, and mineral soil carbon, we contend that most of the uncertainty in our estimate of total pyrogenic emissions arises from uncertainty in combustion of these pools.

[24] In the case of litter and duff, we are reasonably confident that our sample means for preburn mass for both that of ultramafic and nonultramafic substrates approach the true Biscuit-area means. Likely, most of the uncertainty arises from the assumption that combustion factors for litter

Table 5. Preburn Fuel Mass and Combustion Factors by Alternative Convention[a]

		Combustion Factor (Fraction Combusted)			
Fuel Category	Fuel Mass, Mg dry mass ha^{-1}	High Severity	Moderate Severity	Low Severity	Unburned Very Low Severity
Trees	263.2	0.08	0.07	0.03	0.00
Snags	15.7	0.18	0.14	0.11	0.01
Shrubs	3.7	0.86	0.66	0.42	0.00
Nonwoody fuel	<0.1	1.00	0.76	0.75	0.70
1 h surface fuels	6.1	1.00	0.76	0.75	0.70
10 h surface fuels	3.1	0.24	0.08	0.04	0.04
100 h surface fuels	3.6	0.79	0.73	0.67	0.62
1000+ h surface fuels	18.6	0.78	0.58	0.61	0.62
Litter	13.0	1.00	0.76	0.75	0.70
Duff	12.8	0.99	0.51	0.54	0.44

[a]Shrubs include all hardwoods <7.6 cm DBH; unlike elsewhere in paper, here litter excludes all woody fragments. Other categories follow the FCCS fuel category definitions.

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Table 6. Pyrogenic Carbon Emissions by Carbon Pool and Burn Severity Class

Forest Carbon Pool	Combusted Carbon, Mg ha⁻ High Severity	Moderate Severity	Low Severity	Unburned Very Low Severity	Fire-Wide[a] Combustion, Tg C
Litter	7.4	5.5	5.8	5.4	1.00-1.24
Duff, soil and roots	8.3	4.2	4.6	3.5	0.79-1.48
Dead wood	4.8	3.1	3.7	2.9	0.72
Live wood and bark	4.1	2.1	3.0	0.4	0.49
Live foliage	4.1	3.7	1.4	0.2	0.43
Total	28.6	18.6	18.6	12.4	3.83

[a]Calculated by weighting the emissions from each bum class by the area of that bum class over the fire perimeter. Ranges shown for litter, duff, and soil reflect uncertainty in parameter estimates as described in text.

and duff computed for each of the 180 plots did not covary with the actual preburn litter and mass. For instance, if conditions were such that ground fuel consumption was moisture-limited, more litter and duff masses may equate to lower fractional combustion due to greater moisture retention. Conversely, if conditions were such that ground fuel consumption was continuity-limited rather than moisture-limited, lower litter and duff masses may equate to lower fractional combustion.

[25] While our estimate of prebum mineral soil carbon (including roots) was crudely based on samples from only three study plots, by far the most uncertain parameter was the presumed depth to which all carbon was combusted below exposed mineral surfaces identified in the inventory data as either "moderately". or "deeply" charred. Our best estimate of 2.0 and 4.0 cm, respectively, was based on the assumption that surface temperatures during the Biscuit in some cases exceeded 700°C (Bormann, personal communication), that soil temperatures during fire attenuate rapidly with depth, and that soil carbon begins to combust at 100°C *[Agee, 1993]*. However, it is also reasonable to believe that soil carbon could have completely combusted to depths of up to 5 cm or that complete combustion never exceeded 2 cm.

[26] To quantify the potential uncertainty stemming from assumptions regarding litter, duff, and mineral soil combustion, we computed an alternative maximum and minimum value for total pyrogenic emissions across the Biscuit. An alternative maximum value of 4.4 Tg was arrived upon by matching the higher litter and duff combustion factors to higher preburn litter and duff masses (i.e., a positive interaction effect), and assigning deep maximum soil C consumption depths of 3 cm and 5 cm for mineral surfaces identified as moderately and deeply charred, respectively. Similarly, an alternative minimum value of 3.5 Tg was arrived upon by matching the higher litter and duff combustion factors to lower preburn litter and duff masses (i.e., a negative interaction effect), and assigning shallow maximum soil C consumption depths of 1 cm and 2 cm for mineral surfaces identified as moderately and deeply charred, respectively. The litter and duff component of the analysis was performed by first identifying the percentile of each combustion record from the entire distribution, then multiplying each litter and duff combustion record by a preburn mass selected from the same percentile of its distribution (for maximum value), and finally multiplying each litter and duff combustion record by a preburn mass selected from the reverse percentile (100-x) of the prebum mass distribution (for minimum value).

[27] Because the combustion data come from a regular sampling scheme, and because the severity map was used only to bin (not measure) combustion factors, the particular burn severity classification used to bin the plots has little influence on our estimate of fire-wide emissions. The effect of burn severity classification on the estimate of fire-wide emissions arises only from potential covariance between burn severity and preburn carbon density. To investigate this source of uncertainty, we computed an alternative estimate of fire-wide emissions using all the same combustion data but treating all plots as a single bum severity class (equation (1) without the i designation). The resulting estimate of fire-wide pyrogenic emissions was different by only 10%. Because any alternative severity classification would likely have more in common with the BAER classification than no classification at all, it is reasonable to assume that the use of an alternative severity classification would result in a discrepancy in total pyrogenic emissions much smaller than 10%.

4. Discussion

4.1. Comparisons With Other Studies

[28] Overall, the combustion factors reported here for litter and duff (0.70—1.00 for litter and 0.40—1.00 for duff depending on fire severity) are similar to those reported or used by others modeling fire emissions. *Wiedinmyer et al.* [2006] used litter combustion factors of 0.8 to 0.9 depending on tree cover when modeling combustion across North America, *Soja et al.* [2004] used litter combustion factors of 0.2 to 1.0 depending on fire severity when modeling combustion across Siberia, and *Michalek et al.* [2000] used combined litter and humus combustion factors of 0.2 to 0.9 depending on fire severity when modeling combustion for a black spruce forest in Alaska.

[29] Our combustion factors for tree stems (<0.01—0.03 for stems >7.6 cm DBH and <0.01—0.71 for stems <7.6 cm DBH, depending on fire severity) are somewhat lower than values commonly used by modelers. *Wiedinmyer et al.* [2006] used a woody fuel combustion factor of 0.30 when modeling high severity combustion across North America, *Soja et al.* [2004] used a tree combustion factor of 0.30 when modeling high severity combustion across Siberia, and *Lu et al.* [2006] used a tree combustion factor of 0.10 for temperate forests of China. While the definition of woody fuel varies among these studies, the application of these combustion factors to the Biscuit Fire would lead to a large overestimation of pyrogenic emissions, in part because a significant portion of the biomass is in large trees that experience very little wood combustion. Notably, the com-

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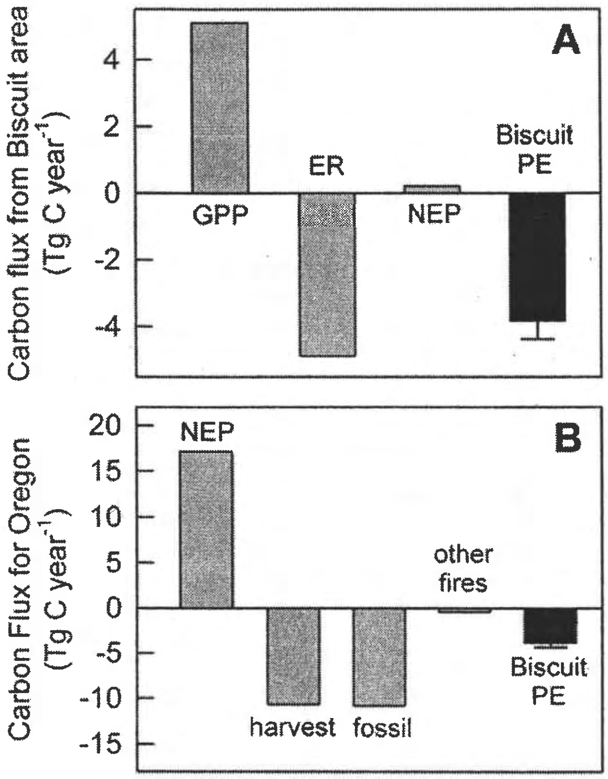

Figure 3. Pyrogenic carbon emissions from the 2002 Biscuit Fire (PE) compared with simulated ecosystem fluxes from (a) the forest present prior to the fire and (b) simulated biome fluxes across Oregon. GPP is Gross Ecosystem Production, NEP is Net Ecosystem Production, ER is total Ecosystem Respiration, and harvest is the sum of both forest product and crop removals. Data for all grey bars are from simulations by *Turner et al.* [2007] averaging the years 1996–2000 except fossil emissions which represent 2000 values from *Blasing et al.* [2004]. Error bar on Biscuit PE covers the upper alternative estimate described in this study.

bustion factors we report here for high severity fire are very similar to those reported for western Washington state, United States, by *Fahnestock and Agee* [1983], who, using no more than expert knowledge, estimated combustion factors to be 0.05, 0.10, 0.75, 0.30, and 0.80 for stems, branches, understory vegetation, dead wood, and forest floor, respectively, in high-severity wildfire.

[30] The latest AP-42, a document used by the U.S. Environmental Protection Agency in estimating air pollution, reports values for fuel loading (mass of fuel typically consumed by wildfire) of 135 and 40 Mg ha^{-1} for Oregon and California forests, respectively. Applying the former of these two values to the Biscuit would yield a total pyrogenic emission of about 14 Tg C (four times that reported in this study). However, applying the latter of these two values to the Biscuit would yield a total pyrogenic emission of about 4 Tg C (just outside our upper estimate). The discrepancy between values for Oregon and California can be traced to *Yamate* [1973], who first compiled fuel loading values for forests of the United States from what were regionally different approaches to estimating forest fuels.

4.2. Utility of Inventory Data

[31] Only through the use of federal inventory data were we able to assess pool-specific carbon losses over an area as large and diverse as that affected by the Biscuit Fire. The addition of fire-related measurements to the normal suite of inventory metrics was done primarily to predict delayed mortality, validate fire behavior models, and monitor the effects of fire on soil. These measurements also proved very useful in making estimates of pyrogenic emissions. The largest limitation to the inventory data used in this study is the absence of preburn litter and duff mass. While one can, as we did, use cover type to assign each plot a regional average value, only by matching observations of combustion to preburn measurements made at the same location can one confidently account for interactions that may exist between preburn mass and the subsequent combustion factor. The addition of litter and duff depth to the standard inventory protocol would go a long way toward improving our ability to estimate carbon losses.

[32] The second most valuable addition to inventory measurement with respect to pyrogenic emissions would be to extend the measurement of dead trees to include those less than 7.6 cm DBH. As determined from data collected in our supplementary plots, a great deal of the mortality and combustion occurred in this smaller size class. If the purpose of postburn inventory is to be expanded to include estimates of pyrogenic emissions of carbon or any another chemical species, it would be highly recommended to modify federal inventory protocols to include assessment of the smaller fire-killed trees. As interest grows in monitoring the effects of and recovery from fire in forests of the western United States, it is likely that federal inventory data will be increasingly relied upon.

4.3. Regional Significance of Biscuit Emissions

[33] One way to consider the importance of pyrogenic emissions from the Biscuit Fire is to compare it to fluxes from the same parcel of ground prior to the fire. As illustrated in Figure 3a, the estimated 3.8 Tg of C released as a result of combustion during the fire is nearly equal to the annual gross primary production, and approximately 18 times the annual net ecosystem production, simulated for an equal area of forest in the same Klamath-Siskiyou ecoregion (data from simulations by *Turner et al.* [2007]). Clearly pyrogenic emissions from a disturbance of this magnitude are an important part of any forest carbon budget. Nevertheless, one must realize that over 60% of the combustion comes from litter, foliage, and small downed wood, all of which are believed to have mean residence times of 10–20 years [*Law et al.*, 2001]. While some fraction of the combusted surface fuels would, without fire, find its way into long-term soil carbon pools, a sizable fraction of the pyrogenic emissions may be thought of as being destined for biogenic emission (i.e., through decay) within 1 to 2 decades with or without fire. Moreover, the proportion of these higher turn-over pools that is combusted should equate to a subsequent reduction in the heterotrophic respiration of these pools until they become recharged by new litter and branch fall. Conversely, carbon pools with

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longer residence times, such as the stems of larger trees, contributed proportionally less to the pyrogenic emissions.

[34] Preliminary calculations suggest that the biomass killed but not combusted by the Biscuit Fire approaches 11 Tg C. As this material decays, the protracted biogenic emissions initiated by the Biscuit Fire should eventually exceed the one-time pyrogenic emission. However, considering that the majority of this fire mortality is in the form of large tree boles, uncertainties in the aerial decay rates of fire-killed trees, the rates at which these trees fall to the ground, and any decompositional effects of charring make it difficult to predict just how this biogenic loss will play out.

[35] Another way to consider the importance of pyrogenic emissions from the Biscuit Fire is to compare this one-time flux to regional fluxes in the same year. As illustrated in Figure 3b, the 3.8 Tg C estimated to have been released by the Biscuit Fire in this study is equal to approximately one third of the 10.8 Tg C reported to be released annually through fossil fuel burning in Oregon *[Biasing et al., 2004]*. Furthermore, our estimate pyrogenic emission from the Biscuit Fire reduces estimates of Net Biome Production in Oregon (Net Ecosystem Production minus timber and crop harvest removals minus average fire emissions) in 2002 by more than half from 6.2 to 2.4 according process simulations made by *Turner et al. [2007]*.

4.4. Future Research

[36] In this paper we estimate the pyrogenic carbon emissions from a particularly large fire in Oregon primarily for the purpose of determining the significance of this historical disturbance event to the carbon balance of the region, but also to explore the utility of federal inventory to do so. Undoubtedly, the most reliable way to extend these computations to future wildfires in the region would be to conduct similar ground measurements on these fires. However, the vast majority of fires in the western United States do not burn large enough to affect an appropriately large number of inventory plots that cover a range of variability in severity and prebum carbon pools. So, in the short term, combustion factors reported here could be applied to other Oregon fires with the assumption that they would be more accurate than other literature values that are derived largely from boreal fires. The observation that BAER severity classification consistently ranked the combustion factors of nearly all 24 prebum carbon pools (Table 3) suggests that it, as well as other classifications derived from remote imagery, may scale combustion factors across fires on comparable forests with acceptable accuracy. Only additional ground studies will be able to confirm this.

[37] One important direction for future work is to better quantify combustive losses from litter, duff, and mineral soil, as this was a primary source of uncertainty in our computations. Especially valuable would be repeated measures of litter and duff mass at the same sample points before and after a fire, as only these studies would reveal any covariance between prebum mass and fraction combusted (a potentially important interactive term not accounted for in equation (1)). Quantifying carbon combustion from mineral soil poses its own challenges. In a meta analysis including eight forest wildfire studies, *Johnson and Curtis [2001]* found substantial variability in the impacts of wildfire on A-horizon carbon content with an overall tendency for this pool to increase following wildfire, which was attributed to additions of charcoal and hydrophobic organic matter. The potential for wildfire to enrich soil carbon, combined with uncertainty surrounding postburn erosion and the sampling error ubiquitous to soil carbon quantification, unfortunately renders the before-after approach for assessing carbon combustion from mineral soil less tractable than it is for litter and duff. For these reasons the mechanistic modeling of soil carbon combustion from fire temperature (as done very crudely in this study) holds more promise than empirical approaches quantifying pyrogenic emissions from forest soils.

[38] Fine scale estimates of fuel loads, fuel consumption, and carbon production across the continental United States, Hawaii and Alaska continue to be improved by the FCCS (Fuel Characteristic Classification System) and fire behavior modles such as Consume 3.0 *[Sandberg et al., 2001; Ottmar et al., 2007]* (http://www.fs.fed.eu/pnw/fera/research/smoke). Future efforts to assess pyrogenic losses will likely be carried out through the use of process-based fire behavior models parameterized with these or similar fuel load layers, and driven by the sort of high precision remote imagery that can measure the intensity and duration of surface energy flux during the course of a wildfire *[Riggan et al., 2004]*. These sophisticated approaches will still require independent estimates of fuel consumption like those that can be provided by prefire and postfire inventory.

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