Universidad Austral de Chile

Facultad de Ciencias Forestales
Bosques Pro Carbono

PROGRAMA CONSULTORES CALIFICADOS
FUNDACIÓN PARA LA INNOVACIÓN AGRARIA
GOBIERNO DE CHILE

INFORME TÉCNICO DE PROPUESTA B-01-007

"FORTALECIMIENTO DE LA CAPACIDAD INSTITUCIONAL DEL PROGRAMA BOSQUES PRO-CARBONO"

INSTITUTO DE MANEJO FORESTAL
FACULTAD DE CIENCIAS FORESTALES
UNIVERSIDAD AUSTRAL DE CHILE

Valdivia, 19 de Noviembre de 2001

I. ORIGEN DE LA PROPUESTA

Desde el año 1997, la Universidad Austral de Chile a través de su Facultad de Ciencias Forestales ha abordado el rol de los bosques de Chile en la búsqueda de alternativas de mitigación de los efectos de cambio climático global, mediante la captura de carbono en los ecosistemas forestales, identificando esta oportunidad como una posibilidad cierta de incorporar ingresos para propietarios forestales y proyectos afines, a través de eventuales ventas de bonos de captura de carbono transables en un mercado mundial en desarrollo.

El Proyecto FONDEF "Medición de la capacidad de captura de carbono en bosques de Chile y promoción en el mercado mundial de carbono" del a mencionada casa de estudios, ha desarrollado la línea de trabajo de investigación, transferencia y asesoría en el tema de medición y monitoreo de captura de carbono en ecosistemas forestales enfocado al sector productivo forestal y científico del país. A partir de los niveles de información alcanzados, se identificó la necesidad de apoyo de aspectos científicos relativos a metodologías de medición y evaluación del carbono en el suelo, tendientes a mejorar los logros del proyecto mencionado. En virtud de los sólidos antecedentes académicos, el avanzado trabajo realizado en la materia de la necesidad detectada, y su participación como coordinador del Grupo de Trabajo IUFRO sobre Captura y Almacenamiento de Carbono, la Universidad Austral de Chile propuso como consultor al **Dr. Robert Jandl**, del Centro de Investigación Forestal de Austria Forstliche Bundesversuchsanstalt,

II. OBJETIVOS DE LA PROPUESTA

El objetivo general de la propuesta ha sido el fortalecer la capacidad institucional para apoyar de mejor forma, futuros desafíos sectoriales en relación con la factibilidad de incorporar proyectos forestales de instituciones y empresas a los mercados de captura de carbono, incrementar la competitividad de esas iniciativas y realizar programas de transferencia.

Los objetivos específicos de esta iniciativa han sido:

a) Incorporar innovaciones tecnológicas en las metodologías de medición de la captura de carbono en unidades forestales, especialmente en relación al balance de carbono en el suelo de los bosques.

b) Conocer los procesos clave y relación con factores del sitio, propiedades químicas y acciones de manejo forestal conducentes a incrementar el potencial de fijación de carbono en el suelo.

c) Acrecentar la interacción con instituciones internacionales, iniciando un proceso de mutua cooperación que contribuya a perfeccionar y armonizar los métodos de monitoreo de secuestro de carbono atmosférico para hacerlos aplicables a la realidad forestal chilena.

d) Apoyar a empresas interesadas en los aspectos técnicos de la contabilidad de los beneficios de carbono, de tal manera de asegurar el cumplimiento de las exigencias de transparencia, exactitud y verificabilidad.

Objetivos de la consultoría, según los términos de referencia.

Brindar apoyo técnico específico a instituciones nacionales en metodologías relativas a mejorar la medición y monitoreo de la captura de carbono atmosférico por ecosistemas involucrados en proyectos forestales.

Transferir soluciones de manejo del bosque para mejorar la capacidad de fijación y conservación sostenible del carbono en el suelo. Ello, favorecerá la competitividad de los proyectos forestales en actual diseño, que incluyan la captura de carbono como vía para mejorar su rentabilidad.

III. EVALUACIÓN DE LA CONSULTORÍA

A. RELACIÓN DE LOS PROFESIONALES INVOLUCRADOS EN EL PROGRAMA DE ACTIVIDADES:

Actividad 1. Visita por parte del Consultor, Dr. Robert Jandl, a predios de la Universidad Austral de Chile, discusión de metodologías de medición de forma práctica en el terreno y capacitación sobre alternativas tecnológicas para medición de balances de C en el suelo.

Actividad 2. Evento de transferencia a empresas forestales nacionales, acerca de la realidad europea en materia de diseño de proyectos forestales de cambio climático y especialmente en manejo de suelos para la fijación de carbono.

Por razones logísticas, las actividades 1 y 2 debieron desarrollarse de manera conjunta, dando lugar a un evento que tuvo lugar en dependencias del Centro Experimental Forestal, en Valdivia, durante la mañana del 16 de Octubre, y que consideró el logro de los objetivos de ambas actividades inicialmente planificadas.

Se realizó una invitación amplia a las empresas forestales de la zona sur del país, afiliadas a la Corporación Nacional de la Madera (CORMA) en sus oficinas central de Concepción y zonal Valdivia, y además se hizo extensiva la invitación a las oficinas regionales de INFOR (Instituto Forestal) con su sede central en Concepción y la sede de Valdivia.

Durante el evento, el Consultor tuvo ocasión de interactuar con las siguientes personas:

1. Dr. Juan Schlatter. Ingeniero Forestal. Profesor titular e Investigador de la Facultad de Ciencias Forestales de la Universidad Austral de Chile.
2. Sr. Felipe Leiva. Ingeniero Forestal. Director Ejecutivo del Centro Experimental Forestal (CEFOR).
3. Sr. Chisen Ley. Ingeniero Forestal. Empresa Forestal Bosques S.A.
4. Sr. Jorge Gayoso. Ingeniero Constructor. Profesor titular e Investigador de la Facultad de Ciencias Forestales de la Universidad Austral de Chile
5. Sr. José Miguel Trabuco. Ingeniero Forestal. Empresa Forestal Valdivia S.A.
6. Sr. Guido Geoffroy. Ingeniero Forestal. Programa Todo-Agro.
7. Sr. Jorge Izaurieta San Juan. Ingeniero Forestal. Contratista particular.
8. Srta. Sigrid Calderón. Ingeniero Forestal. Proyecto FONDEF "Medición de la capacidad de captura de carbono en bosques de Chile y promoción en el mercado mundial de carbono".
9. Sr. Javier Guerra. Ingeniero Forestal. Programa Producción Forestal y Medioambiente, Universidad Austral de Chile.
10. Sr. Diego Alarcón. Ingeniero Forestal. Programa Producción Forestal y Medioambiente, Universidad Austral de Chile.

Actividad 3. Presentación de innovaciones metodológicas europeas en el ámbito de la medición y monitoreo del secuestro del carbono atmosférico.

Esta actividad se llevó a cabo en la Sala Federico Saelzer de la Facultad de Ciencias Forestales de la Universidad Austral de Chile, el día 16 de octubre, a partir de las 16:00 hrs, y hasta las 17:50 hrs.

Los asistentes sumaron 43 personas, comprendiendo cuerpo docente de la Universidad Austral de Chile, el equipo de investigación de medición y monitoreo de captura de carbono en bosques Chilenos, además de Estudiantes tanto de Pregrado como de Magister de la Facultad de Ciencias Forestales.

La lista de los asistentes es la siguiente:

Nombre	RUT
Aburto Droppelmann, Yasna	13524710-3
Andrade Gallardo, Cesar Alejandro	13322992-2
Araya Quezada, Luis Jonathan	13633764-5
Armesto Soto, Sebastian	13261170-K
Brunet Hebel, Jean Pierre	13521566-K
Cañulef Cañulef, Roberto	13404144-7
Carcamo Gonzalez, Miguel	13116910-8
Donoso Gasti, Felipe	10689101-K
Epuyao Leal, Jaime Patricio	13161669-4
Farias Herrera, Aldo Miguel	14262191-6
Follert Emhart, Alejandro Daniel	13321491-7
Fresard Brandstatter, Carlos	13457621-9
Galvez Sandoval, Jose Miguel	9421722-9
Gambi Moreira, Bruno	12721324-0
Garrido Gonzalez, Manuel Alejandro	14269503-0
Gayoso Aguilar, Jorge	
Gayoso Morelli, Paola	12084822-4
Gonzalez Ojeda, Carlos Alberto	13117136-6
Gutiérrez Carreño, David Alejandro	14332168-1
Hechenleitner Vega, Francisco	12024311-K
Herrera Figueroa,Victor	12994214-2
Higuera Sanhueza, Enrique	10665300-3
Hornig Acuña, Jorge Andres	12033484-0
Hueitra Quilaqueo, Richard Luis	11918628-5
Maragaño Burgos, Manuel Eduardo	9309860-9
Marquez Ortiz, Julio Cesar	12750249-8
Minte Gomez, Antonio Enrique	13167224-1
Morales Medina, Luis Alejandro	13318456-2
Muñoz Moya, Roberto Arturo	
Olivares Cerpa, Gonzalo Javier	13021099-6
Paredes, Gonzalo	
Parra Casanelli, Rodrigo Ivan	13731843-1
Poo Barrera, Cesar Andres	12592605-3

Ramirez Rebolledo, Rodrigo	13281945-9
Romero Peña, Patricio Fabian	13710033-9
Saavedra Catalan, Jorge Waldo	12865541-7
Sagardia Parga, Rodrigo Oscar	12430334-6
Schultz Oyarzun, Felipe Andres	14346111-4
Silva Jerez, Javier Andres	12993525-1
Stange Hofmann, Rodrigo Daniel	13523749-3
Vergara Asenjo, Gerardo Enrique	12390359-5
Villegas Barria, Jose Ricardo	12935236-1
Witt Fuchs, Mathias	10031893-8

En el marco del objetivo de la actividad 3, se llevaron a cabo, además, las siguientes reuniones:

a) Reunión con el Dr. Renato Grez, Docente e Investigador del Laboratorio de Suelos Forestales del Instituto de Silvicultura de la Facultad de Ciencias Forestales de la Universidad Austral de Chile.

b) Reunión con Rodrigo Sagardia, del Programa de Producción Forestal y Medioambiente, del Instituto de Manejo Forestal de la Facultad de Ciencias Forestales de la Universidad Austral de Chile.

Actividad 4. Reunión de cooperación con el equipo técnico del proyecto nacional en curso "Medición de la capacidad de captura de carbono en bosques de Chile y promoción en el mercado mundial de carbono", de la Universidad Austral de Chile en conjunto con el Instituto Forestal.

Esta actividad se desarrolló en dos fracciones:

a) Reunión con el módulo "Mercado de Captura de Carbono", del Proyecto FONDEF, el día 17 de Octubre, 15:00 a 16:30 hrs, en la Sede Valdivia del Instituto Forestal (INFOR).

   Los asistentes fueron:
   1. Sr. Jorge Urrutia. Ingeniero Forestal, docente del Instituto de Manejo Forestal, Facultad de Ciencias Forestales de la Universidad Austral de Chile.
   2. Sr. Jorge Cabrera. Ingeniero Forestal. Director de la Sede Valdivia del Instituto Forestal (INFOR).
   3. Srta. María Rosa Alzamora. Docente Instituto de Manejo Forestal, Facultad de Ciencias Forestales de la Universidad Austral de Chile.

b) Reunión con el módulo "Metodologías de Medición y Monitoreo de Captura de Carbono en los Ecosistemas Forestales Chilenos", del Proyecto FONDEF, el día 17 de Octubre, de 11:30 a 13:00 hrs, en dependencias de la Universidad Austral de Chile.

Los asistentes fueron:

1. Sr. Jorge Gayoso. Ingeniero Constructor. Profesor titular e Investigador del Instituto de Manejo Forestal de la Facultad de Ciencias Forestales de la Universidad Austral de Chile.
2. Sra. Bastienne Schlegel. Ingeniero Forestal. Programa de Cooperación Técnica Internacional Chile-Chiapas, México.
3. Srta. Sigrid Calderón. Ingeniero Forestal. Proyecto FONDEF "Medición de la capacidad de captura de carbono en bosques de Chile y promoción en el mercado mundial de carbono".
4. Sr. Javier Guerra. Ingeniero Forestal. Programa Producción Forestal y Medioambiente, Universidad Austral de Chile.
5. Sr. Diego Alarcón. Ingeniero Forestal. Programa Producción Forestal y Medioambiente, Universidad Austral de Chile.
6. Sr. David Gutiérrez. Ingeniero Forestal. Programa Producción Forestal y Medioambiente, Universidad Austral de Chile

Actividad 5. Colaboración al desarrollo del Simposio Internacional organizado por la Universidad Austral de Chile, denominado "Medición y Monitoreo de la Captura de Carbono en Ecosistemas Forestales" durante los días 18 a 20 de Octubre de 2001, en Valdivia.

Los participantes de esta actividad totalizaron 52 profesionales Chilenos y 42 internacionales, en su mayoría de Latinoamérica. El listado de los asistentes se presenta a continuación.

Asistentes Nacionales. A las actividades del Simposio, se congregaron profesionales de diversas instituciones del quehacer científico nacional tanto de Universidades como Institutos Tecnológicos y Centros de Formación, al igual que distintas empresas representativas de la amplia gama del rubro forestal productivo nacional:

	Apellido	Nombre	E-mail	Institución/Empresa	Ciudad
1	Ahumada Fonseca	Isaac	iahumada@tutopia.com	Univ. Iberoamericana de Ciencia y Tecnología	Santiago
2	Alarcón	Diego	dalarcon@uach.cl	Universidad Austral de Chile	Valdivia
3	Arrué Figueroa	Karina	karruef@entelchile.net	Universidad de Chile	Santiago
4	Astete	Jaime	jastetea@netscape.net	Universidad Mayor	Santiago
5	Bertín	Jaime	jbertin@inacap.cl	INACAP Valdivia	Valdivia
6	Cabrera	Jorge	jcabrera@infor.cl	Instituto Forestal	Valdivia
7	Calderón	Sigrid	carbono@uach.cl	Universidad Austral de Chile	Valdivia
8	Cartes Quilodrán	Cristian	cristiancartes@hotmail.com	Univ. Iberoamericana de Ciencia y Tecnología	Santiago
9	Castro	Mauricio	maucastr@udec.cl	Universidad de Concepción	Concepción
10	Celis	Margarita	mcelis@millalemu.cl	Forestal Millalemu	Cabrero
11	Contreras	Luis	contreras.luis@entelchile.net	Forestal Savia Ltda. Punta Arenas	Santiago

12 Espinosa	Cristian	cespinosa@cipres.utalca.cl	Universidad de Talca	Talca
13 Gayoso	Jorge	jgayoso@uach.cl	Universidad Austral de Chile	Valdivia
14 Gayoso	Sylvana	sgayoso@entelchile.net	Universidad Austral de Chile	Valdivia
15 George	Laurent	conafonf@interweb.cl	ONF Conosur - Proyecto CONAF	Temuco
16 González	Carlos	cagonzalez@arauco.cl	Bioforest S.A.	Concepción
17 Grandjean	Jorge	jgrandjeano@yahoo.com	Universidad Austral de Chile	Valdivia
18 Guerra	Javier	jguerra@uach.cl	Universidad Austral de Chile	Valdivia
19 Gutierrez	David	dgutierrez@uach.cl	Universidad Austral de Chile	Valdivia
20 Henriquez	Manuel	mhenriquez@conaf.cl	CONAF, Coyhaique	Coyhaique
21 Husch	Bertram	infora1@entelchile.net	INFORA Estudios Ltda	Santiago
22 Huss	Elke	ehuss@entelchile.net	Universidad Austral de Chile	Valdivia
23 Ipinza	Roberto	ripinza@infor.cl	Instituto Forestal, Coyhaique	Coyhaique
24 Ivanovich	Juan	jivanovi@conaf.cl	CONAF, XII región	Punta Arenas
25 Jara	Miguel	majara@interweb.cl	CONAF, Temuco	Temuco
26 Le Moigne Oliva	Jean	jotadroad@hotmail.com	Univ. Iberoamericana de Ciencia y Tecnología	Santiago
27 Lira	Valentina	vlira@formin.cmpc.cl	Forestal Mininco S.A.	Santiago
28 López Leiva	Domingo	dalopez@hotmail.com	INACAP Valdivia	Valdivia
29 Lucero	Hugo	hlucero@conaf.cl	CONAF, Provincial Malleco	Angol
30 Maldonado	Adrián	amald@mixmail.com	Universidad de la Frontera	Temuco
31 Marfull	Francisco	fmarfull@puc.cl	Universidad Católica de Chile	Santiago
32 Meza	Andrés	ameza@conaf.cl	CONAF	Santiago
33 Moreno	Paulo	pmoreno@uach.cl	Universidad Austral de Chile	Valdivia
34 Muñoz	Roberto	ramunoz@uach.cl	Universidad Austral de Chile	Valdivia
35 Neuen-schwander	Aquiles	aquilesn@fia.gob.cl	FIA	Santiago
36 Nieto	José Antonio	marceloni80@hotmail.com	INACAP Valdivia	Valdivia
37 Núñez	Patricio	pnunez@ufro.cl	Universidad de la Frontera	Temuco
38 Ortega	Alicia	aortega@uach.cl	Universidad Austral de Chile	Valdivia
39 Pugin	Alejandra	apugin@info.com	Universidad Austral de Chile	Cañete
40 Sagardía	Rodrigo	rsagardia@uach.cl	Universidad Austral de Chile	Valdivia
41 Sanhueza	José Eduardo	j.sanhueza@mi-mail.cl	CONAMA	Santiago
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43 Searle	Juan Pedro	jpsearle@conama.cl	CONAMA	Santiago
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46 Torres	Juan Carlos	juancarlostorres@entelchile.net	Universidad Austral de Chile	Valdivia
47 Torres	Richard	rtorres@conaf.cl	CONAF	Santiago
48 Urrutia	Jorge	jurrutia@uach.cl	Universidad Austral de Chile	Valdivia
49 Vargas	Ricardo	rvargas@conaf.cl	CONAF, IX región	Temuco
50 Vega	Mario	mhvega@uach.cl	Universidad Austral de Chile	Valdivia
51 Villalobos	José Miguel	jomikel@ctcinternet.cl	Universidad Austral de Chile	Valdivia
52 Zamora	Nelson	nzamora@uach.cl	Universidad Austral de Chile	Valdivia

Asistentes internacionales. La actividad del Simposio Internacional convocó a científicos y profesionales de las disciplinas relacionadas a actividades forestales de las principales instituciones de diversos países, con una mayor concentración de toda Latinoamérica, relacionadas con las metodologías de medición y monitoreo de captura de carbono asociado a actividades de manejo forestal:

	Apellido	Nombre	E - mail	Institución	Ciudad	País
1	Acosta	Miguel	acostamm@colpos.mx	Colegio de Post Graduados - INFFAR	Montecillo	México
2	Brown	Sandra	sbrown@winrock.org	Winrock International	Arlington	USA
3	Caballero	Danice	danjcecab@yahoo.com	U.A.G.R.M	Sta Cruz de la Sierra	Bolivia
4	Callo - Concha	Daniel	dancacon@hotmail.com	Universidad Autonoma de Chapingo	Texcoco	México
5	Calmon	Miguel	m_calmon@uol.com.br	The Natural Conservancy do Brasil (TNC)	Curitiba/Paraná	Brasil
6	Cardoso Vale	Luiz Carlos	deorei@terra.com.br	Del Rey Servicios de Ingeniería Ltda.	Belo Horizonte	Brasil
7	Catpo Chuchón	Jorge Enrique	19970172@lamolina.edu.pe	Universidad Nac. Agraria La Molina	Lima	Perú
8	Cony	Mariano	mcony@lab.cricyt.edu.ar	IADIZA, Argentina	Mendoza	Argentina
9	de Jong	Bernardus	bjong@sclc.ecosur.mx	Colegio Frontera del Sur, Villa Hermosa	Villa hermosa	México
10	Díaz	Arnoldo	arnol@ar.inter.net	Bosque Andino (Servicios forestales y vivero)	Esquel	Argentina
11	Etchevers	Jorge	jetchev@colpos.mx	Colegio de Post Graduados	Montecillo	México
12	Franco	Ximena	ximenaf@impsat.net.co	Fundación Natura	Bogotá	Colombia
13	González	Germán Luis	germang@chavin.rcp.net.pe	Minga Perú	Lima	Perú
14	Guarderas Vela	Jaime Eduardo	tropimaderas@andinanet.net	Tropimaderas S.A.	Quito	Ecuador
15	Gutmanis	Daina	dainag@terra.com.br	Instituto de zootecnia UNESP	Sao Paulo	Brasil
16	Jara	Luis Fernando	profafo1@uio.satnet.net	Profafor S.A. Ecuador	Quito	Ecuador
17	Loguercio	Gabriel	gloguer@ciefap.cyt.edu.ar	CIEFAP	Esquel	Argentina
18	Lopera Arango	Gabriel	gajalopera@mixmail.com	Universidad Nacional de Colombia	Copacabana	Colombia
19	López	Yamel	yamel@emcali.net.co	Universidad Nacional de Colombia	Cali	Colombia
20	Loza - Balbuena	Isabel	ilozabalbuena@hotmail.com	School of Forestry, Christchurch	Christchurch	Nueva Zelanda
21	Manfrinato	Warwick	wm@merconet.com.br	CENA - Universidad de Sao Paulo	Sao Paulo	Brasil
22	Masera	Omar	omasera@ate.oikos.unam.mx	Univ. Nacional Autónoma de México	Michoacán	México
23	Mayen	José Octavio	jmayen@uach.cl	Magister Universidad Austral de Chile		Honduras
24	Merle	Carolina	caro.merle@netcourrier.com	ONF Internacional	Bogotá	Colombia

25 Monreal	Carlos	monrealc@em.agr.ca	Agriculture and Agri-Food	Ottawa, Ontario	Canadá
26 Montes	Rubén	ramontes@usb.ve	Universidad Simón Bolivar	Caracas	Venezuela
27 Mora Galán	César	cytasa@hotmail.com	CYTASA S.A.	Asunción	Paraguay
28 Morales	Manuel	mmuburgos@hotmail.com	Magister Universidad Austral de Chile		Bolivia
29 Muñoz	Thelmo	tmuñoz@uach.cl	Magister Universidad Austral de Chile		Bolivia
30 Navar	José de Jesus	jnavar@ccr.dsi.uanl.mx	Facultad de Ciencias Forestales, AUNL	Linares	México
31 Orrego	Sergio Alonso	saorrego@perseus.unalmed.edu.co	Univ. Nacional de Colombia - Depto Cs. Forestales	Medellín	Colombia
32 Ortiz	Edgar	eortiz@itcr.ac.cr	Instituto Tecnológico de Costa Rica	Cartago	Costa Rica
33 Oyhant-Çabal	Walter	woyha@mgap.gub.uy	Ministerio de Ganadería, Agricultura y Pesca	Montevideo	Colombia
34 Picado	Freddy	comunac@ibw.com.ni	Ministerio del Ambiente y Recursos Naturales	Managua	Nicaragua
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36 Saéz	Adrián	saeza@infovia.com.ar	Adrían Saez Ingeniería	Buenos Aires	Argentina
37 San José	José	jsanjose@oikos.ivic.ve	Instituto Venezolano de Investigaciones Científicas	Edo. Miranda	Venezuela
38 Sarasola	Mauro	msarasola@bariloche.inta.gov.ar	Instituto nacional de Tecnología Agropecuaria	Bariloche	Argentina
39 Schlegel	Bastienne	bcschlegel@netscape.net	Programa de Cooperación Técnica Internacional	Chiapas	México
40 Tiepolo	Gilberto	carbono@spvs.org.br	SPVS	Curitiva - Paraná	Brasil
41 Tórrez	Bernardo	comunac@ibw.com.ni	Ministerio del Ambiente y Recursos Naturales	Managua	Nicaragua
42 Vásquez Vilchez	Carlos Alberto	ramayang@hotmail.com	U.N. Agraria La Molina	Lima	Perú

B. ANTECEDENTES RELACIONADOS CON LA OPERACIÓN. DESCRIPCIÓN Y ANÁLISIS DE LA METODOLOGÍA EMPLEADA, PROCEDIMIENTOS EMPLEADOS Y RESULTADOS OBTENIDOS

Actividad 1. Visita a predios de la Universidad Austral de Chile, discusión de metodologías de medición de forma práctica en el terreno y capacitación sobre alternativas tecnológicas para medición de balances de C en el suelo.

Actividad 2. Evento de transferencia a empresas forestales nacionales, acerca de la realidad europea en materia de diseño de proyectos forestales de cambio climático y especialmente en manejo de suelos para la fijación de carbono.

Estas dos actividades se llevaron a cabo en un solo evento que se realizó en los predios del Centro Experimental Forestal, desde las 09:30 y hasta las 13:00 hrs.

El encuentro se inició con la observación de calicatas de suelos forestales con una descripción y caracterización de las principales propiedades físicas y químicas, junto con un reseña histórica de los usos de los suelos forestales en la región centro-sur de Chile, dirigida por el Dr. Juan Schlatter y el profesor Jorge Gayoso, como introducción de la problemática del uso de los suelos para una mejor comprensión del Consultor acerca de la realidad local, sirviendo a la vez de base para la discusión de la captura de carbono en suelos forestales.

Se inició la discusión acerca de las metodologías de medición de manera práctica en terreno, dando a conocer las diferentes formas de medición de flujos de C en el suelo, en relación con las variables que determinan la acumulación de carbono en los suelos, posibles de contabilizar en proyectos forestales que incluyan captura de carbono como opción de mejora de balance económico.

Frente a la observación de las diferentes situaciones de caracterización físico-químico, en los horizontes minerales y orgánicos, ante diferentes esquemas de manejo y especies forestales establecidas, se discutieron opciones innovativas de manejo que permitan al propietario o empresa optimizar la captura de carbono en los suelos. Se abordaron de igual forma aspectos relacionados con consecuencias históricas de impactos en suelos forestales debido a procesos de amasado y compactación por usos agropecuarios anteriores, y por técnicas inapropiadas de preparación de sitio para plantaciones forestales, que hayan redundado en eliminación o disminución de materia orgánica, lo cual hace desfavorable el balance de C para proyectos que consideren el almacenamiento de carbono en los bosques.

EL Consultor tuvo ocasión de transmitir a los presentes diversa información correspondiente a la experiencia de la red de monitoreo de suelos forestales de Europa central, con un mayor énfasis en las experiencias de Austria y Alemania, quienes han desarrollado diversas alternativas para favorecer la acumulación de C en los ecosistemas forestales, basándose en una productividad mayor de los rodales,

permitiendo mayor acumulación en la biomasa. Asimismo, la exclusión del empleo del fuego como técnica de preparación de sitios, ha representado un plus a los sistemas forestales para conservar C. Respecto de las propiedades fisico-químicas de los suelos forestales del centro sur de Chile, se discutieron las fortalezas para una mayor consideración de almacenamiento de carbono en los suelos, basados en técnicas de conservación de la capa orgánica y la elección de especies y proveniencias adecuadas a cada sitio.

El Dr. Jandl además presentó las variables relevantes en el diseño de proyectos forestales con énfasis en captura de carbono, según se han dado las decisiones del Protocolo de Kyoto, que representan las bases de eventuales alternativas de negocio para proyectos forestales.
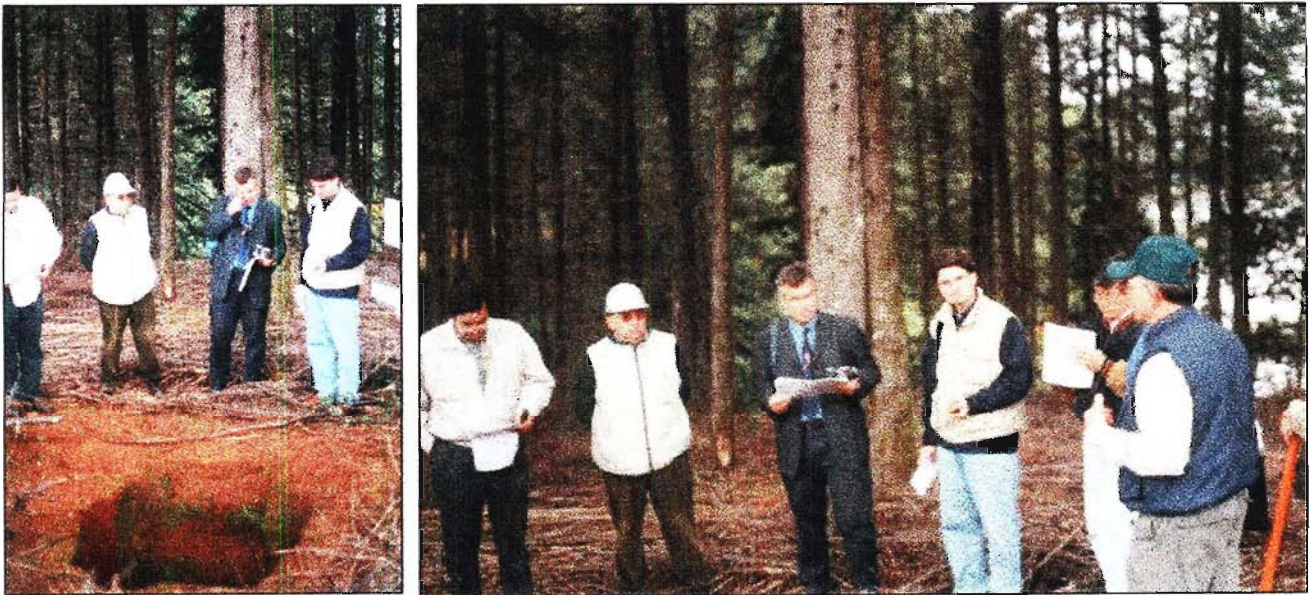
Figura 1. Los asistentes discuten las propiedades físicas y químicas de los suelos de aptitud forestal en el Centro Experimental Forestal. En las fotografía principal, de izquierda a derecha: don Jorge Izaurieta, don Jorge Gayoso, de la Universidad Austral de Chile, Dr. Robert Jandl, El Sr. Felipe Leiva, de CEFOR, El Sr. Chisen Ley, de Bosques S.A. y el Dr. Juan Schlatter, de la Universidad Austral de Chile.

Figura 2. Los asistentes recorren dependencias del Centro Experimental Forestal, donde el Dr. Schlatter muestra regeneración natural de *Pseudotsuga menziesii* con protección del manto orgánico, favoreciendo el almacenamiento de C en los bosques.
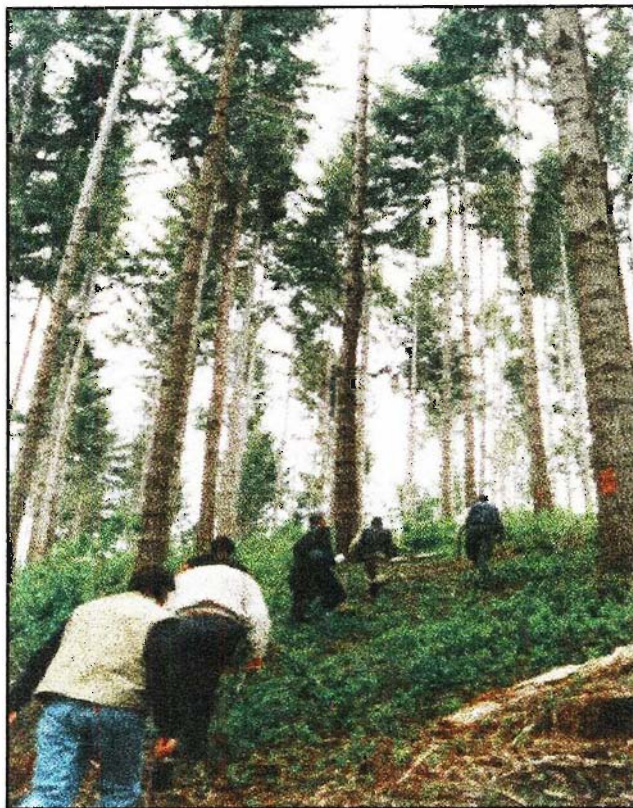
Figura 3. Los asistentes discuten acerca de la inconveniencia del empleo de la quema controlada como método de preparación de sitios, desde el punto de vista de la acumulación de carbono comprobable para proyectos de captura de carbono.

Actividad 3. Presentación de innovaciones metodológicas europeas en el ámbito de la medición y monitoreo del secuestro del carbono atmosférico.

Esta actividad se llevó a cabo en la Sala Federico Saelzer de la Facultad de Ciencias Forestales de la Universidad Austral de Chile, el día 16 de octubre, a partir de las 16:00 hrs, y hasta las 17:50 hrs.

El Consultor expuso a través de una conferencia, los alcances del sector forestal europeo actual, las últimas investigaciones y opciones de manejo de los bosques y suelos de aptitud forestal. Una copia de láminas de la presentación se adjunta en el capítulo de información entregada por el consultor.

En el marco del objetivo de la actividad 3, se llevaron a cabo, además, las siguientes reuniones:

a) Reunión con el Dr. Renato Grez, Docente e Investigador del Laboratorio de Suelos Forestales del Instituto de Silvicultura de la Facultad de Ciencias Forestales de la Universidad Austral de Chile.

b) Reunión con Rodrigo Sagardia, del Programa de Producción Forestal y Medioambiente, del Instituto de Manejo Forestal de la Facultad de Ciencias Forestales de la Universidad Austral de Chile.

Durante estos encuentros, además, se trataron temas específicos sobre las particularidades de las propiedades químicas de los suelos del centro-sur de Chile. Como resultado de estas actividades, el Dr. Jandl colaboró en la aproximación a la resolución de problemas de productividad de sitios, dando directrices generales de las distintas alternativas existentes para abordarlo, asociados a opciones de almacenamiento de C en los suelos:

La hipótesis con que se trabajó enuncia que: el suelo y en consecuencia la productividad de las plantaciones forestales sufren debido a actividades como la cosecha, preparación de sitio y algunos tipos de manejo silvícola.

De acuerdo al doctor Jandl, las cuatro posibilidades existentes para la investigación de estos efectos en la situación particular y en la incidencia del almacenamiento de C, reside en las:

- investigaciones químicas de los suelos: para esta situación indica que la erosión lleva a pérdidas de nutrientes si la capa superficial del suelo es deslizada. Esta pérdida se expresa en bajos contenidos de carbono, nitrógeno y potasio, sin embargo son indicadores que hay que utilizar con cautela, especialmente en sitios fértiles, dónde el impacto de dichas técnicas es menor. Otro factores a considerar al respecto es si el agua accesible constituye una limitante para la absorción de nutrientes de los vegetales. Señala además que las alteraciones en el crecimiento de los vegetales debido a déficit de nutrientes son de naturaleza esporádica, y que

hay que tomar en cuenta la adaptabilidad de las especies vegetales para desarrollarse en suelos pobres.

- investigaciones físicas de los suelos: dónde el factor más común que afecta la productividad es la compactación del suelo, existiendo muchos métodos para medirlo. Agrega que la redundancia en este tipo de estudios es considerable (Burger, 1998).

- investigaciones microbiológicas: Enuncia la importancia de la mineralización de materia orgánica como el motor del ciclo de nutrientes, indicando que si la actividad de los microbios disminuye, la oferta de nutrientes baja asimismo. De las recomendaciones para realizar medir esta actividad, señala que un método a bajo costo es medir la respiración con "soda-lime" (ref. 1-4), y cuyo principio es la captura del CO_2 derivado de la respiración del suelo en forma de Na_2CO_3.

- investigación del crecimiento de los árboles en el laboratorio: la propone como una alternativa para probar distintos tratamientos al suelo de forma controlada, y dónde todos los factores del sitio salvo el tratamiento permanecen constantes.

De igual manera el Dr. Jandl, dio orientación sobre la manera en que habría llevar a cabo la investigación, según restricciones como la significancia de los factores a medir, sus costos asociados y la escala de tiempo a la que van asociados a este problema específico. Al respecto señaló que:

- según la opinión de James Burger de Virginia Tech, la mejor medida tras integrar otras cualidades del suelo, es el empleo de la variable "volumen de agua accesible'". Este corresponde al volumen del agua del suelo entre dos distintos puntos de la función pF, por ejemplo entre el punto de saturación y el punto de marchitez permanente. El valor final de esta medida es el tiempo durante el cual el suelo es capaz de sustentar a las plantas. El método requiere de establecer la función pF para todas las capas de los suelos, lo cual se puede llevar a cabo a bajo costo aún cuando implica una gran cantidad de trabajo en laboratorio.

- medir la resistencia del suelo contra la rotación de un objeto también puede ser una medida efectiva, pero al precio de una precisión menor y de dejar de lado otros factores que son relevantes. Sin embargo, tiene la ventaja de poder realizarse a bajo costo y con rapidez.

- *el empleo de sitios de referencia o parcelas permanentes es* ideal es el monitoreo de algunos sitios por varios años. Como esto dificulta la finalización rápida del proyecto, propone como solución compensar el espacio por tiempo, es decir, medir muchos sitios en diferentes fases antes y después del impacto. En este caso los sitios deben poseer características similares en cuanto a su topografía, tipo de suelo y ubicación geográfica.

- *respecto de los costos enfatiza que,* en general, las investigaciones químicas son caras, existiendo un conjunto mínimo de datos a recopilar para obtener un análisis concluyente del estado químico del suelo. Sugiere además que si se está lidiando con daños que afectan únicamente las características físicas del suelo, la importancia de los análisis químicos es menor.

- *de la evolución de los impactos a los sitios a través del tiempo señala que,* inmediatamente después del impacto los daños del suelo pueden ser grandes, pero según la naturaleza e intensidad del daño, el suelo puede regenerar tras un horizonte de tiempo discreto. Para poder comparar los resultados, los sitios a evaluar deben constar de un tiempo de rotación típico. Uno de los elementos sobre los cuales hizo énfasis para no inducir a error respecto de otros estudios, es que cualquier conclusión que se haga sobre los efectos de cosecha sobre un sitio, debe incluir la descripción del tipo de sitio que para el cual se aplicó.

Las siguientes son algunas referencias aconsejadas por el Consultor es esta actividad:

1. Nelson T. Edwards, 1982, The use of soda-lime for measuring respiration rates in terrestrial systems, Pedobiologia, 23, 321-330
2. S. Pongracic, M.U.F. Kirschbaum, R.J. Raison, 1997, Comparison of soda lime and infrared gas analysis techniques for *in situ* measurement of forest soil respiration, Canad J Forest Research 27, 1890-1895.
3. Richard D. Bowden, Knute J. Nadelhoffer, Richard D. Boone, Jerry M. Melillo and Jason B. Garrison, 1993, Contribution of aboveground litter, belowground litter, and root respiration to total soil respiration in a temperate mixed hardwood forest, Canad J Forest Research 23, 1402-1407.
4. Mark Nay and Kim G. Mattson and Bernard T. Bormann, 1994, Biases of chamber methods for measuring soil CO_2 efflux demonstrated with a laboratory appartus, Ecology 75, 2460-2463.
5. P.A. Stevens, D.A. Norris, T.G. Williams, S. Hughes, D.W.H. Durrant, M.A. Anderson, N.S. Weatherley, M. Hornung and C. Woods, 1995, Nutrient losses after clearfelling in Beddgelert forest: a comparison of the effects of conventional and whole-tree harvest on soil water chemistry, Forestry 68, 115-131.
6. Chris E. Johnson, Arthur H. Johnson, Thomas G. Huntington, Thomas G. Siccama, 1991, Whole-tree clear-cutting effects on soil horizons and organic-matter pools, Soil Science Society America Journal 55, 497-508 (2 papeles)
7. James A. Burger and Daniel L. Kelting (autores), Mary Beth Adams, Kilaparti Ramakrishna Eric A. Davidson (editores), 1998, Soil quality monitoring for assessing sustainable forest management, in "The contribution of soil science to the development of and implemenation of criteria and indicators of sustainable forest management", Soil Science Society America, Special Publication 53, 17-52, Soil Science Society of America, Madison, Wisconsin

Actividad 4. Reunión de cooperación con el equipo técnico del proyecto nacional en curso "Medición de la capacidad de captura de carbono en bosques de Chile y promoción en el mercado mundial de carbono", de la Universidad Austral de Chile en conjunto con el Instituto Forestal.

Esta actividad se desarrolló en dos fracciones:

1. Reunión con el módulo "Mercado de Captura de Carbono", del Proyecto FONDEF, el día 17 de Octubre, 15:00 a 16:30 hrs, en la Sede Valdivia del Instituto Forestal (INFOR).

2. Reunión con el módulo "Metodologías de Medición y Monitoreo de Captura de Carbono en los Ecosistemas Forestales Chilenos", del Proyecto FONDEF, el día 17 de Octubre, de 11:30 a 13:00 hrs, en dependencias de la Universidad Austral de Chile.

En ambas reuniones, el Consultor ayudó al equipo técnico en definir diversos aspectos técnicos referentes a la medición y monitoreo de la captura de carbono en los ecosistemas forestales, así como el pensamiento actual en instituciones líderes a nivel mundial, como la IUFRO. Los temas a discutir, abarcaron las áreas de :

- Alcances de los sucesos actuales de avance del Protocolo de Kyoto
- Tipos de Proyectos en el Mecanismo de Desarrollo Limpio (MDL) y factores-fortaleza del sector forestal Chileno para la entrada de productores nacionales al proceso.
- Rol de los Suelos Forestales en el almacenamiento de Carbono, comparación de las metodologías de medición de los programas europeo, en contraste con los avances de las metodologías desarrolladas en el marco del proyecto FONDEF.
- Experiencias de la Red de Parcelas de Monitoreo de Carbono en Austria, restricciones de medición, importancia de la continuidad de las mediciones en el confiabilidad de la contabilidad del carbono almacenado ante prácticas de manejo silviculturales.
- Detalles de los Proyectos de Monitoreo de Suelos en Austria.
- Posibilidades para transferencia de Experiencias en Control de Suelos en Chile.
- Montos de Reducciones de Emisiones para Austria según Anexo B del Protocolo de Kyoto.
- Posibilidades de Desarrollo de Proyectos en Chile en el marco del MDL como inversión de la Industria Austríaca.
- Análisis y estudios de la formulación de proyectos y el acuerdo de incluir este tema en la Task-Force de IUFRO.

Actividad 5. Colaboración al desarrollo del Simposio Internacional organizado por la Universidad Austral de Chile, denominado "Medición y Monitoreo de la Captura de Carbono en Ecosistemas Forestales" durante los días 18 a 20 de Octubre de 2001, en Valdivia.

Como coordinador de los temas de discusión de la medición y monitoreo de carbono en los suelos forestales, dentro del Simposio Internacional, Robert Jandl discutió aspectos de algunos trabajos relevantes sobre investigación en materia de medición y monitoreo de almacenamiento de carbono en suelos forestales.

Los Autores Warwick Manfrinato; Marisa De Cássia; Carlos Cerri; Martial Bernoux; Carlos Pellegrino, del Centro de Energía Nuclear y Agricultura (CENA) de la Universidad de São Paulo, Brasil, presentaron los resultados de una investigación denominada "Medición de la variabilidad espacial y temporal del carbono del suelo con el uso de los isótopos estables, en una transición bosque-pradera en el Estado del Paraná, Brasil". Durante esta presentación, se logró discutir procedimientos de cuantificación de carbono en los suelos de uso forestal, y sobre las ventajas del empleo de análisis espaciales en la evaluacion de flujos de C en la matriz edáfica de suelos forestales.

Por su parte, los autores Juan E. Schlatter y Víctor Gerding, ambos docentes e investigadores de la Facultad de Ciencias Forestales de la Universidad Austral de Chile, presentaron los resultados de su investigación denominada "Balance de carbono en suelos arenosos con plantaciones de *Pinus radiata*, en Chile central", que dio luces a una discusión amplia de opciones silvícolas de plantaciones en Chile, y los cambios en los flujos de C asociados a los suelos sometidos a uso forestal.

Ante estas presentaciones, el Consultor discutió los aspectos importantes a considerar en las mediciones de carbono edáfico y una vez más aportó con las opciones silviculturales para mejorar la captura de C en los ecosistemas forestales y plantaciones forestales.

El Consultor desarrolló además, una Conferencia con el tema "Medición de tendencias en el tiempo del almacenamiento de carbono en el suelo", mediante la cual explicó a los asistentes los alcances de investigación necesarios para un efectivo monitoreo de carbono en los suelos, con ejemplos de las experiencias de Austria y otros países de Europa Central.

Figura 4. Colaboración del Consultor en el Simposio Internacional de Medición y Monitoreo de Carbono en Ecosistemas Forestales. En la fotografía de izquierda a derecha: El Sr. Carlos Weber, Director Ejecutivo de la Corporación Nacional Forestal, El Sr. Manfred Max-Neef, Rector de la Universidad Austral de Chile, El Sr. Jorge Gayoso, docente e investigador de la Facultad de Ciencias Forestales de la Universidad Austral de Chile, y el Dr. Robert Jandl, Consultor calificado.

C. CONCLUSIONES Y RECOMENDACIONES PROPUESTAS

La participación del Consultor en las actividades desarrolladas concluye en:

Se ha creado conciencia de la importancia de desarrollar planes que consideren alternativas de manejo silvicultural que propendan al almacenamiento del C en los suelos de uso forestal en las empresas, propietarios e investigadores del área forestal.

Se ha reconocido la plena factibilidad del sector forestal chileno de incorporarse al mercado en desarrollo de venta de bonos de captura de carbono, a través de los ecosistemas forestales.

Se han reconocido que el principal desafío de considerar el suelo en el almacenamiento de carbono corresponde a que se cuenta con grandes stocks que cambian muy poco en relación a los cambios en el uso del suelo o cambios en el manejo. Este asunto será una constante que deberá incluirse mediante el desarrollo de procedimientos específicos en los futuros proyectos forestales que deseen demostrar una adicionalidad en la captura de carbono para eventualmente transarse en los mercados.

Se reconoce que la dinámica del carbono en los suelos es un tema de investigación de la edafología y únicamente el hecho de medir los aportes e ingresos, no es suficiente como para entender realmente los fenómenos en su conjunto. En el caso de considerar proyectos de desarrollo de captura de carbono, se hace necesario profundizar en las mediciones. Sin embargo, el conjunto de información nueva que se necesita generar en las ciencias forestales no estará inmediatamente disponible a corto plazo, por lo tanto no se debería supeditar al avance del Protocolo de Kyoto, ya que no se sabe con un nivel suficiente de detalle lo que esta ocurriendo en la dinámica de carbono en el suelo, por lo que su investigación debe seguir adelante.

El inicio de la formulación de proyectos que pretendan considerar el almacenamiento de carbono en los diferentes compartimentos del ecosistema, incluyendo el suelo, debe considerar como punto inicial el que es posible el registro de los stocks de carbono en el suelo y que la metodología para su medición es un asunto tecnológicamente resuelto. No obstante, lo importante para los proyectos mencionados, es registrar los cambios que suceden en los contenidos de carbono en cada fracción del ecosistema, y el logro de un monitoreo confiable de dichos cambios reviste un nivel de dificultad mayor. En este sentido, la posibilidad técnica más fácil es asociar los cambios de stock a los cambios de uso del suelo, que por tratarse de procesos drásticos, permite cuantificar los cambios ocurridos de contenido de carbono en los suelos.

Por otro lado, lo que pasa con el cambio de uso del suelo en un proyecto es que se puede medir el carbono a través de los efectos que se producirá en cuanto a

aumentos o conversiones en la cubierta forestal. Si se considera procesos para mejorar el ecosistema, entonces el cambio de contenido de carbono va a ser como una curva logística: será más rápido al inicio y vamos a poder medir los cambios, pero a medida que el ecosistema se va aproximando a su punto de equilibrio va a ser cada vez más difícil medir los cambios. Entonces tenemos que pensar en cuales proyectos vamos a seleccionar para que se puedan medir estos cambios en el uso del suelo. Por ejemplo, en los proyectos basados en un sistema de manejo sostenible de bosques, va a ser muy difícil que se puedan incorporar la fracción donde se obtenga ganancia de carbono en el suelo, porque el diferencial de acumulación será muy difícil de medir en estos proyectos.

El desafío futuro estará en vincular el manejo forestal sostenible con proyectos de captura de carbono para llevar a cabo proyectos de investigación más integrados. En ese sentido, las instituciones internacionales, dentro de las cuales cuenta IUFRO, están velando por que el carbono se analice no sólo desde un punto de vista específico y aislado, sino que pase a constituir una variable más de una estrategia para el manejo sostenible de los bosques. Debido a esto, los esquemas de investigación que involucren medición y monitoreo del carbono en la fracción suelo de los ecosistemas forestales, involucrando tanto plantaciones forestales como proyectos de manejo de bosques nativos, debe considerarse siempre dentro de un contexto integrado de manejo sostenible de bosques.

D. ACTIVIDADES DE DIFUSIÓN Y TRANSFERENCIA REALIZADAS.

La totalidad de las actividades ya expuestas en el programa de fortalecimiento de la capacidad institucional corresponden *per se,* a actividades de difusión y transferencia, y como tales se han explicado en los capítulos anteriores.

Actividades 1. Visita a predios de la Universidad Austral de Chile, discusión de metodologías de medición de forma práctica en el terreno y capacitación sobre alternativas tecnológicas para medición de balances de C en el suelo.

Actividad 2. Evento de transferencia a empresas forestales nacionales, acerca de la realidad europea en materia de diseño de proyectos forestales de cambio climático y especialmente en manejo de suelos para la fijación de carbono.

Estas dos actividades se llevaron a cabo en un solo evento que se realizó en los predios del Centro Experimental Forestal, desde las 09:30 y hasta las 13:00 hrs.

Actividad 3. Presentación de innovaciones metodológicas europeas en el ámbito de la medición y monitoreo del secuestro del carbono atmosférico.

Esta actividad se llevó a cabo en la Sala Federico Saelzer de la Facultad de Ciencias Forestales de la Universidad Austral de Chile, el día 16 de octubre, a partir de las 16:00 hrs, y hasta las 17:50 hrs.

Actividad 4. Reunión de cooperación con el equipo técnico del proyecto nacional en curso "Medición de la capacidad de captura de carbono en bosques de Chile y promoción en el mercado mundial de carbono", de la Universidad Austral de Chile en conjunto con el Instituto Forestal.

Esta actividad se desarrolló en dos fracciones:

a) Reunión con el módulo "Mercado de Captura de Carbono", del Proyecto FONDEF, el día 17 de Octubre, 15:00 a 16:30 hrs, en la Sede Valdivia del Instituto Forestal (INFOR).

b) Reunión con el módulo "Metodologías de Medición y Monitoreo de Captura de Carbono en los Ecosistemas Forestales Chilenos", del Proyecto FONDEF, el día 17 de Octubre, de 11:30 a 13:00 hrs, en dependencias de la Universidad Austral de Chile.

Actividad 5. Colaboración al desarrollo del Simposio Internacional organizado por la Universidad Austral de Chile, denominado "Medición y Monitoreo de la Captura de Carbono en Ecosistemas Forestales" durante los días 18 a 20 de Octubre de 2001, en Valdivia.

E. INFORMACIÓN ENTREGADA POR EL CONSULTOR

Publicaciones entregadas por el consultor:

Newmann, M., Schnabel, G., Gärtner, M., Starlinger, F., Fürst, A., Mutsch, F. Englisch, M., Smidt, S. **Jandl, R.** y Gartner, K. 2001. Waldzustandsmonitoring in Österreich. Ergenbnisse der Intensivbeobachtungsflächen (Level II). Forstliche Bundesversuchsanstalt Wien. Waldforschungszentrum. Forest Condition Monitoring in Austria. Results of the Permanent Observation Plots (Level II).

Jandl, R., Kopezski, H., Bruckner, A., Hager, H. 2001. Forest soil chemistry and mesofauna 20 years after an amelioration fertilization. Restoration Ecology, in submitted on september 2001 for revision.

Neumann, M. 1996. Österreichesches Waldschaden-Beobachtungssystem. Austrian Forest Damage Monitoring System. Contributione to the Fourth Vienna Workshop, November 23rd, 1995. Bundesministerium für Land und Forstwirtschaft.

Jandl, R. 1998a. Modeling Processes in Forest Soils - problems, simplifications and caveats. Ecological Engineering 10: 33-51.

Jandl, R., Kopeszki, H., et Glatzel, G. 1997a. Effect of a dense Allium ursinum (L.) Ground Cover on Nutrient Dynamics and Mesofauna of a Fagus sylvatica (L.) Woodland. Plant and Soil 187: 245-255.

Jandl, R. et Sletten, R. 1999. Mineralization of Forest Soil Carbon: Interactions with metals. Journal of Plant Nutrition Soil Science 162: 623-629.

Jandl, R., et Sollins, P. 1997. Water Extractable Soil Carbon in Relation to Belowground Carbon Dynamics. Biology and Fertility of Soils 25: 196-201.

Jandl, R., Glatzel, G., Katzensteiner, K., et Eckmuellner, O. 2001. Amelioration of magnesium deficiency in a Norway spruce stand (Picea abies) with calcined magnesite. Water, Air, and Soil Pollution, 125: 1-17.

Jandl, R., Kilian, W., Englisch, M., et Mutsch, F. 1999b. Bodenkundliche Untersuchungen auf den Diagnoseprofilflächen. Mitteilungen der Forstlichen Bundesversuchsanstalt 172: 139-218.

Jandl, R., Starlinger, F, Englishc, M., Herzberger, E. 2001. Long-term effects of a forest amelioration experiment. Paper accepted for Canadian Journal Forest Research.

Weiss, P., Schieler, K., Schadauer, K., Radunsky, K. et Englisch, M. 2000. Die Kohlenstoffbilanz des Österreichischen waldes und betrachtungen zum Kyoto-Protocoll. Fostliche Bundesversuchsansalt und Austrian Federal Environment Agency.

Además, el Consultor entregó la siguiente información en formato electrónico:

Gaudinski, J., Trumbore, S., Davidson, E. et Zheng, S. 2000. Soil carbon cycling in a temperate forest: radiocarbon-based estimates of residence times, sequestration rates and partitioning of fluxes. Biogeochemistry 51: 33-69.

Foster, J. 2001. Statistical power in forest monitoring. Forest Ecology and Management 151: 211-222.

Liski, J., Karjalainen, T., Pussinen, A., Nabuurs, G., Kauppi, P. 2000. Trees as carbon sinks and sources in the European Union. Environmental Science and Policy 3: 91-97.

Johnson, D., Sogn, T., Kvindesland, S. 2000. The nutrient cycling model. Forest Ecology and Management 138: 91-106.

Quideau, S., Anderson, M., Graham, R., Chadwick, O., Trumbore, S. 2000. Soil organic matter processes: characterization by ^{13}C NMR and ^{14}C measurements. Forest Ecology and Management 138: 19-27.

Sector forestal de Europa

Puntos de vista: escandinavo -- atlántico -- alpine -- mediterráneo

Tópicos

- Peculiaridades

  derecho forestal
  estructura de los propiedades - Austria
  escala pequeña, beneficios bajos
  2 especies: picea / haya
  historia del manejo

- deterioro de los bosques

  Abeto blanco (*Abies alba*): azufre
  Roble (*Quercus sp.*): falta agua
  Abeto rojo (*Picea abies*): ??

- investigaciones recientes

  exceso de nitrógeno
  conversión de plantaciones de picea entre bosques naturales

Derecho forestal

- Se aplica a todos los involucrados (propiedad pública, privada, industria)
- La preocupación mayor es la sostenibilidad
- Todos los bosques accesibles salvo durante cortes (desde 1975)
- Restricciones para mountain biking, esquiar, recolectar hongos, bayas
- Es fácil convertir áreas en bosques, pero es difícil roturar
- Comunicar infestaciones de insectos, tomar contramedidas
- Ningunos lodos residuales; !permitido en tierras de uso agrario!
- Regeneración de bosques en 10 años después del corte
- Montes protectores donde es racional -- restricciones para involucrados

Guardabosques forman un grupo distinto que se preocupa por los bosques

Pocas universidades forestales -- curricula similares -- Enseñanza cubre todo, *i.e.* ecología, economía, technología, genética, defender torrentes y

Bosques de Austria

- 3.9×10^6 ha bosques (~47% de la área)

- crecimiento / tasa de cortes

- espécies majores

- aumento de la área boscosa (6% desde 1992-96)

53 % agricultores (interés moderado)
15 % gobierno (Oesterreichische
        Bundesforste - bosques reales)
32 % industría (nobleza)

58% picea (Picea abies)
6.5% pino (P silvestris)
2.4% abeto (Abies alba)
9% haya (Fagus sylvatica)
2% roble (Quercus petrea)
pocos especies sobrevivieron la época glacial,
Alpes como barrera Oeste-Este

m^3 tallo / ha

Kleinwald Betriebe ÖBF

Zuwachs/ha   Nutzung/ha

Baumartenmischungen

Fichtenbestände
♦ andere Nadelholzreinbestände
Nadel-Laubmischbestände
♦ Laub-Nadelmischbestände
♦ Laubholzreinbestände
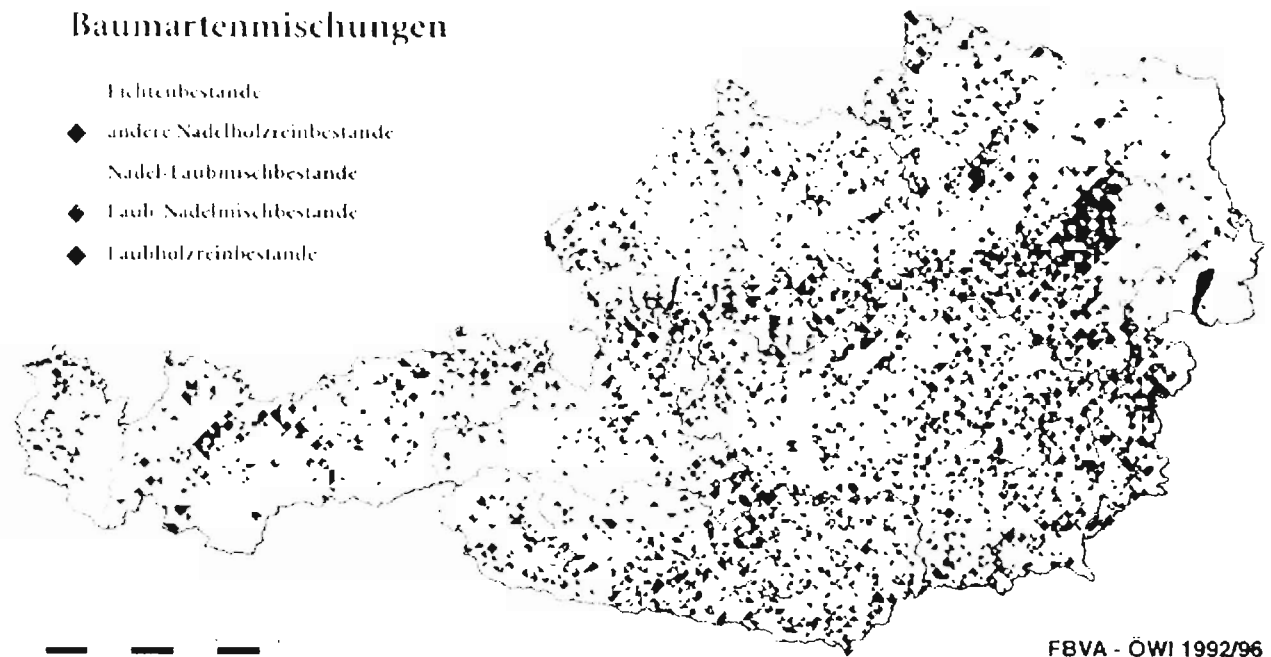
FBVA - ÖWI 1992/96

Flächen der natürlichen Waldgesellschaften

♦ Lärche - Zirbe   ♦ Buche

Fichte   Eiche

Fichte - Tanne   ♦ Andere

♦ Fichte - Tanne - Buche
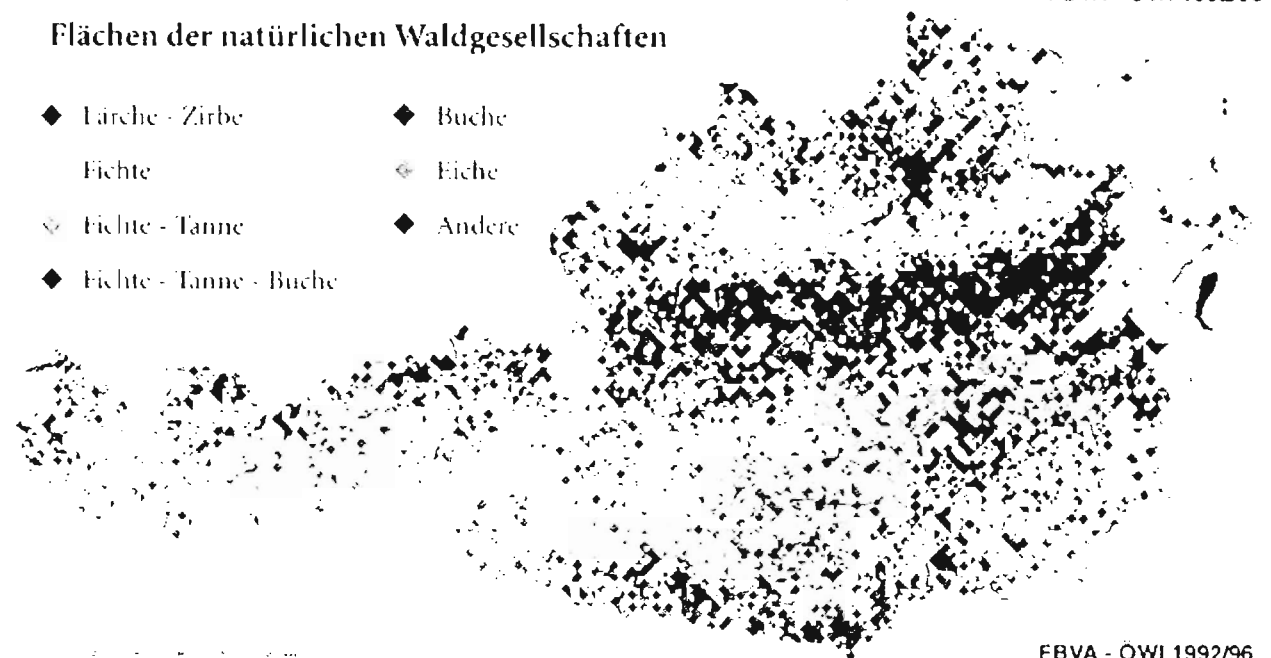
FBVA - ÖWI 1992/96
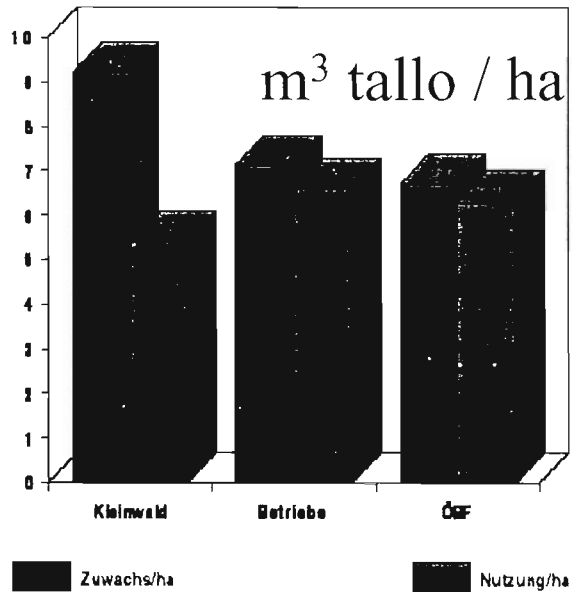
Manejo a escala pequeña

- área de talas como máximo 2 ha (permiso requerido para talas más largas)
- talas selectivas *('Plenterwald')*
- objetivo es la permanencia de la cobertura)
- campesinos: uso doméstico; leña que arde, tejado de los establos
- ganancia del sector forestal en Europa < 2% (peor que cuentas de ahorrar)

Historia 1: Edad de piedra - edad de bronce

- 13kaños aC, después de la época glacial - praderas, habitadas por cazadores & colectores - invasión con arbustos (sauce, avelana, pino, abedul, aliso, roble, ulmo); capacidad de áreas boscosas baja, por eso asentamientos al lado de ríos, lagunas; bajo impacto de los hombres sobre los bosques

- Edad de piedra, agricultura & ganadería; quema prescrita (ceniza fértil, altos rendimientos transientes), shifting cultivation; bosques como obstáculos; árboles que toleran sombra (picea, haya)

- 3kaños aC Edad de bronce: sal; alta demanda de madera para montes y estufas (carbón de madera); red de carreteras para comercio

- 1.5kaños aC migraciones barabarios; depoblación; regeneración de la cobertura boscosa

Source: G Glatzel, 1994: Leben mit dem Wald: Österreichs Wälder im Wechsel der Zeiten. In W. Morawetz (ed): Ökologische Grundwerte in Österreich - Modell für Europa? ÖAkWiss, Wien, 289-303.

Pollendiagramm
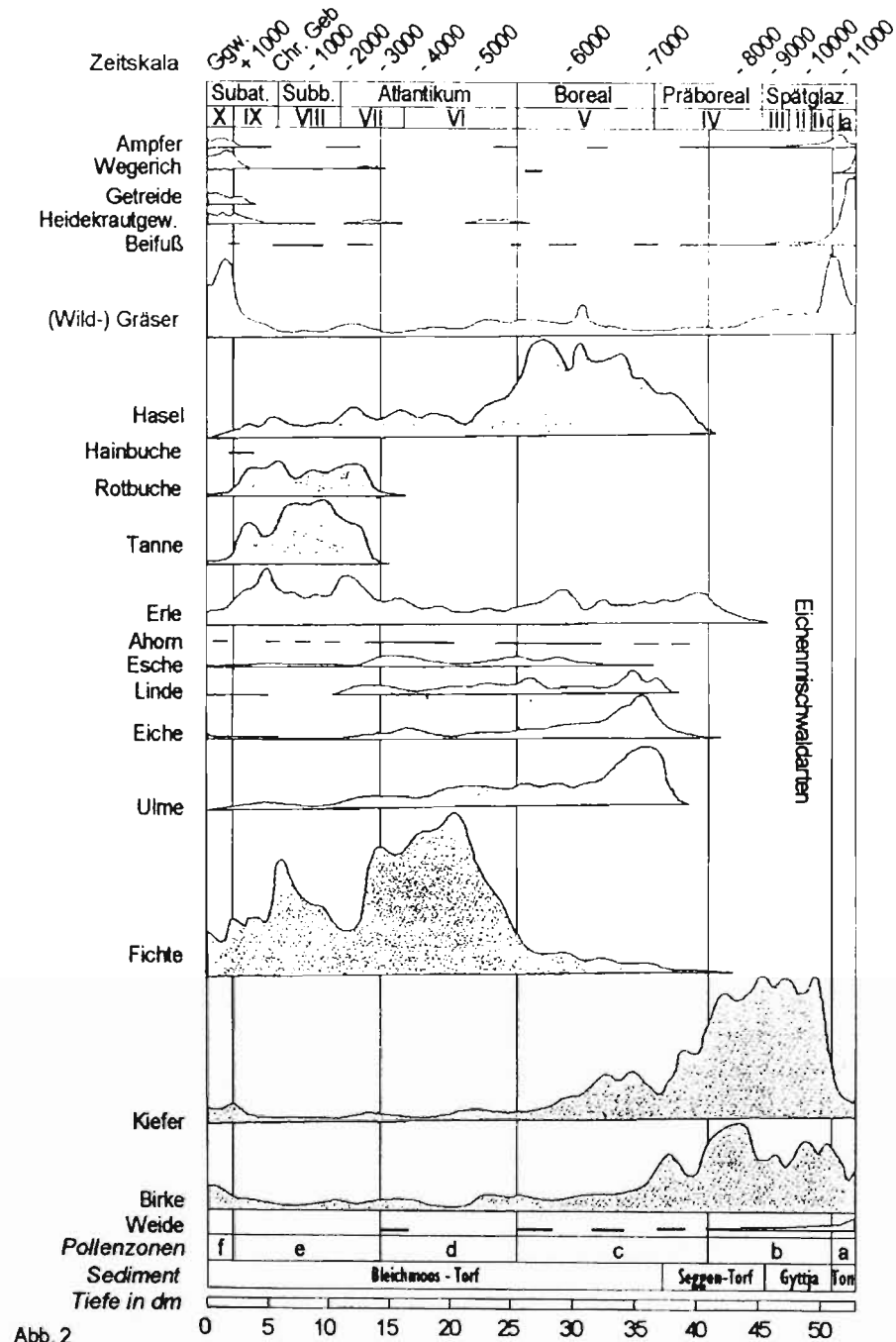
Abb. 2

Historia 2: Edad media (600 - 1600), Edad moderna

- monasterios -> nueva ola de deforestación; cobertura de bosca más baja que hoy
- ‚Ordenaciones del bosque (derecho)‘ para mantener los bosques para nobleza (minería)
- ‚época glacial pequeña‘, hambre, enfermedads, guerras disminuyen la población, los bosques vuelven
- Revolución industrial (~1770) amenaza a los bosques; altísima demanda para leña que arde; fuentes de energía: ríos, carbón de madera; industría de vidrio depende de la madera / evaporación de sal / ciudades crecientes
- Alimento de la población creciendo: rastrillo de la hojarasca, deforestación, agroselvicultura en las talas, bosques como praderas
- bosques abiertos (pino) en los Alpes, prados en Inglaterra y el Norte de Alemania Hausegger, 1861: ‚*..dudoso si todavía hay bosques naturales en Europa ...*‘
- deforestación en los Alpes lleva a torrentes y deslizamienteos (Suiza: experimentos clásicos en ‚Sperbelgraben vs Rappengraben‘ como inicio de las investigaciones de torrentes y aludes)
- reforestaciones con semillas de origen desconocido
- primeras facultades de selvicultura

Source: G Glatzel, 1994
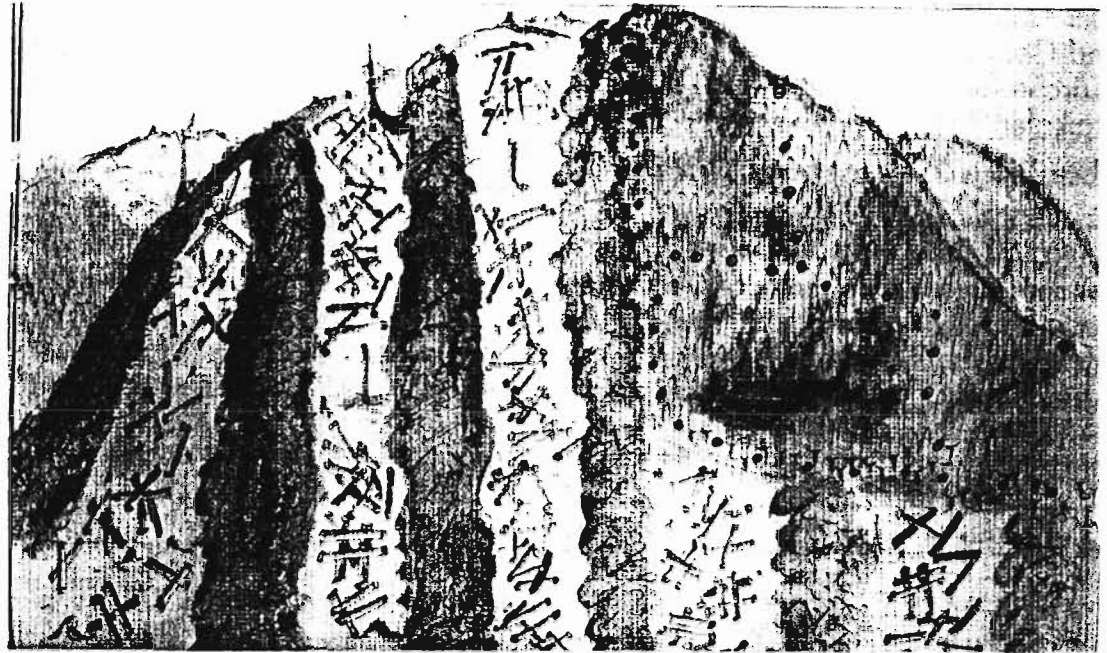
# Historia 3: Carbón salva y después amenaza los bosques

- cerca de 1850: carbón como energía despreciado --> sucio
- Justus von Liebig promueve abonos minerales con nitrógeno
- promoción de picea

- polución del aire a causa de procesos de combustión después de 1950 carbón más barato contiene 3% azufre

Dudas recientes:

altas deposiciones de nitrógeno amenazan la biodiversidad; extinción de comunidades acidófilas?

demasiados corzos

cambios en la hidrología regional

Schwerr
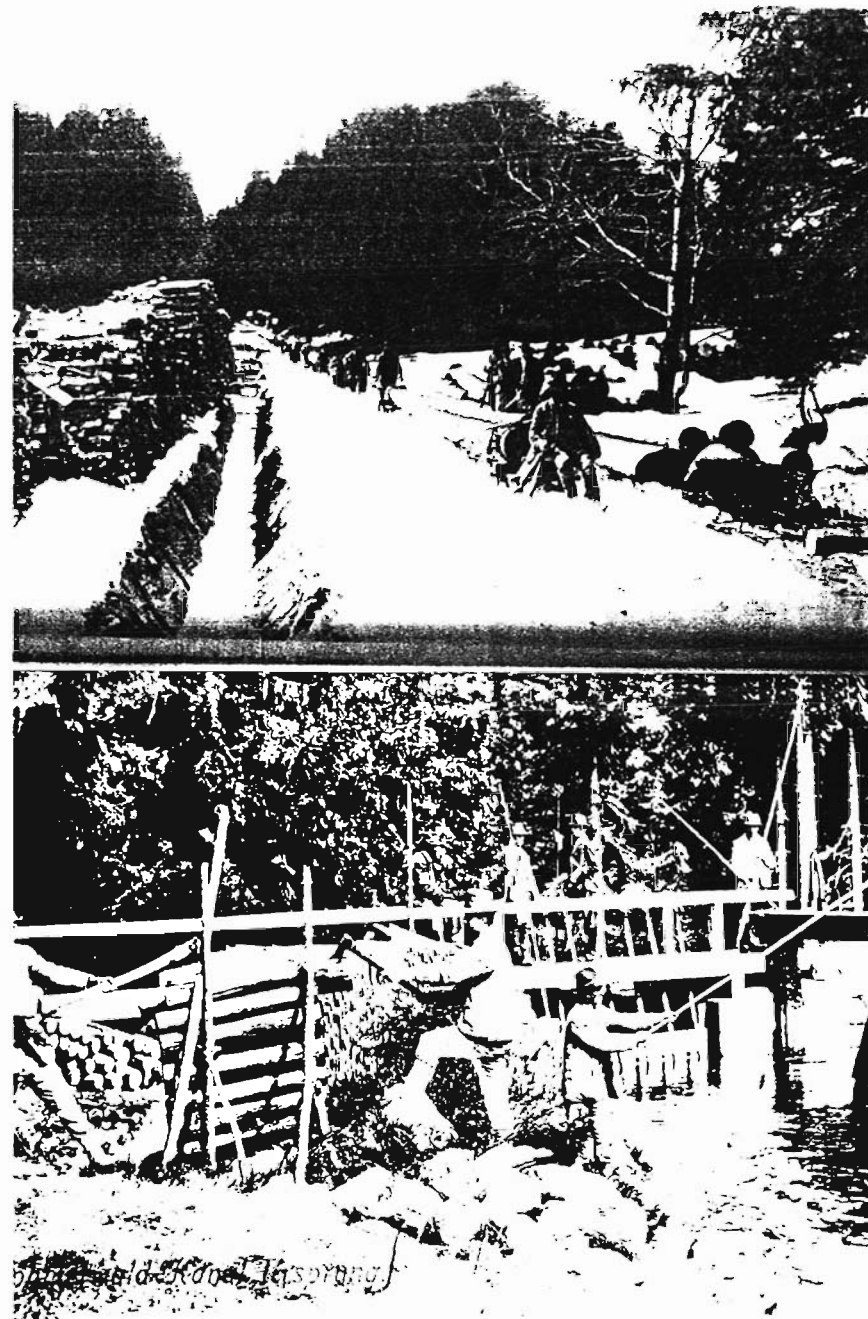
Historía 4: Pérdidas de nitrógeno y acidificación de los suelos debido al rastrillo de la hojarasca ~1850

Área boscosa W-Austria	614.000 ha
60% accesible para rastrillo	370.000 ha
Cantidad del material rastrillado	$4.85 \ 10^6 \ m^3$
... cada ha	$13.1 \ m^3 ha^{-1} yr^{-1}$
... peso seco	
(14 mg N g^{-1}, ANC 1.4 mmol g^{-1})	$1500 \ kg \ ha^{-1} yr^{-1}$
Exportación de N	13-27 $kg \ N \ ha^{-1} yr^{-1}$
Acidificación del suelo	1.8-2.8 $kmol \ ha^{-1} yr^{-1}$

Source: G Glatzel: Historic forest use and its possible implications to recently accelerated tree growth in Central Europe, EFI Proceedings 27, 1999, 65-74

Deterioro de los bosques: dependencia de la masa de acículas y el crecimiento

- altas deposiciones de azufre amenazan abetos; quedan pocos
- defoliación de picea
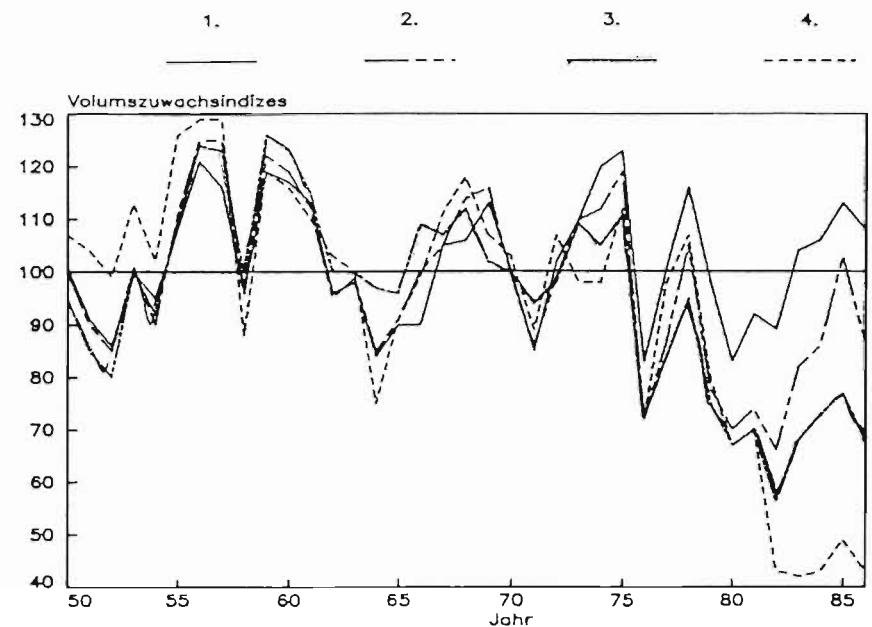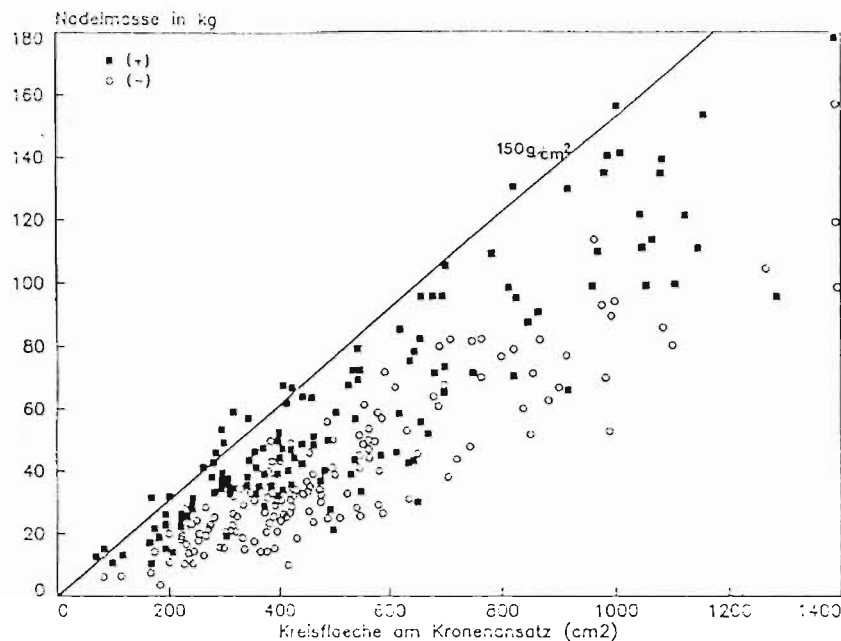
Deterioro de los bosques 2: acidificación de los suelos

• aceptado: móviles llevan al exporto de nutrientes;
crítico en sitios de larga historia del manejo de la tierra;
ejemplo: deficiencia de magnesio; descoloración de las acículas
• rechazado: daños directos debido a la toxicidad del alumino
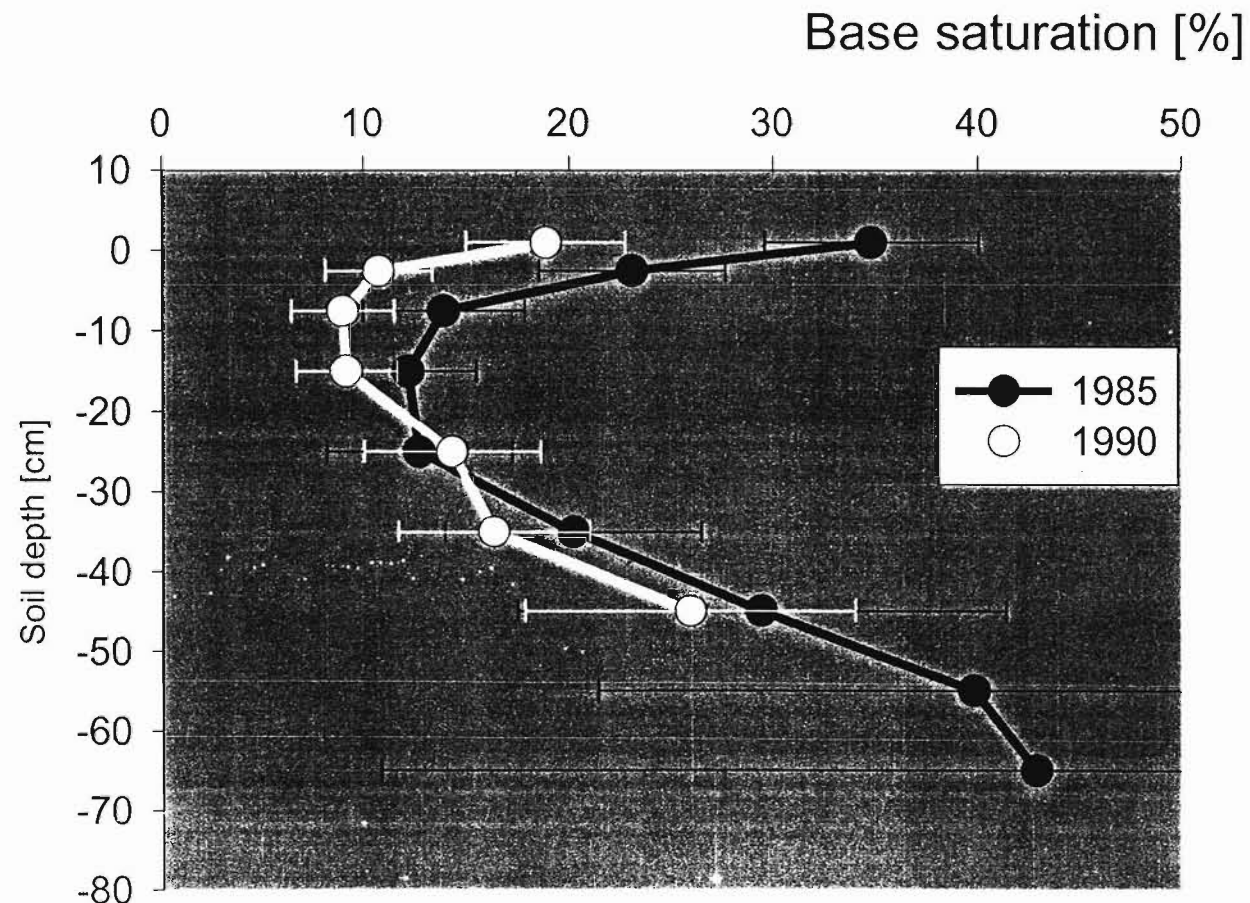
Nitrógeno excesivo (1)

Tasa de deposición de N varia ; 10-40 N kg ha^{-1} yr^{-1};
pre-industrial ~ 5 kg N ha^{-1} yr^{-1}
Ecosistemas forestales acostumbradas a baja provisión de N
demasiado N sin precedencia
Nitrógeno y agua como factores limitantes para los bosques
europeos

Beneficios: altas tasas de crecimiento
Preocupación: falta de equilibrio de nutrientes (Mg)
altas concentraciones en el agua freática
alta disposición para infestaciones de insectos
pérdida de sitios infertiles --- biodiversidad

Asimilación de nutrientes como consecuencia (no la razón) del
crecimiento;
bajas concentraciones de nutrientes - síntomas de deficiencia

Nitrógeno excesivo (2)

height [m]

Source: Kenk *et al.*, 1991, KFZ PEF 82
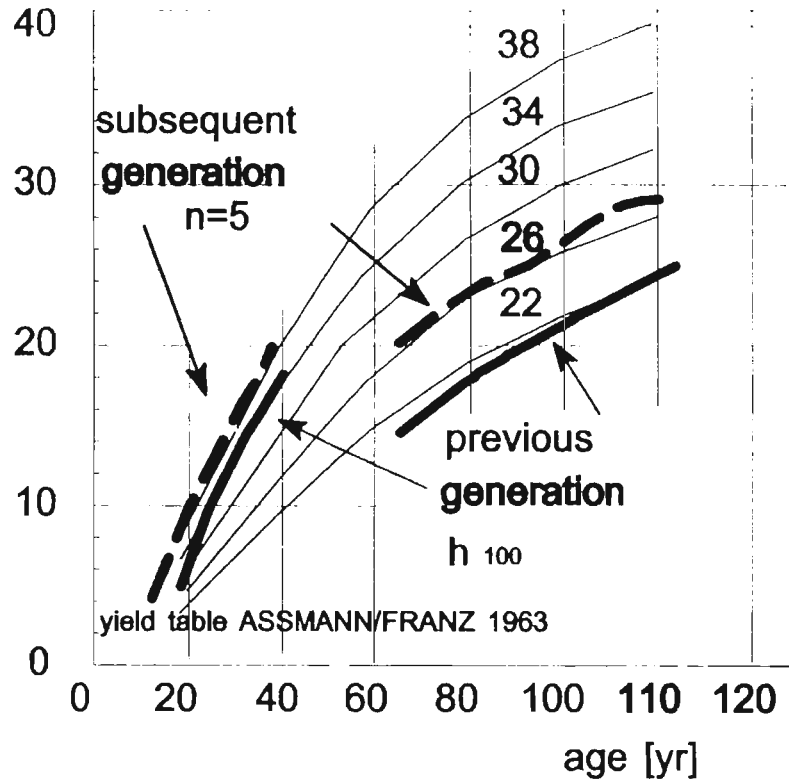
Razones:

nitrógeno addicional

bias   cambio del uso de la tierra
rastrillo
Tablas de crecimiento con error
error sistemático de los inventarios forestale
temperatura (largo periodo del crecimiento
aumento del CO_2

Bosques mixtos (1)

Ventaja
- distribución de riesgos (p.e. insectos)
- estabilidad de los rodales (especies auxiliares para picea
- utilización eficiente de nutrientes (raízes en capas diferentes)
- cobertura boscosa sostenible (hay fase de decaimiento en picea)

CRITICOS
involucrados:
contento con picea; poco manejo, alta tasa de crecimiento
Industria de sierra:
tecnología perfecta para picea
económico

Bosques mixtos (2)

establecimiento de latifoliadas en rodales de picea es difícil
dar abono tiene sentido
minimizar la densidad de corzos

faltan experimentos con observaciones largas
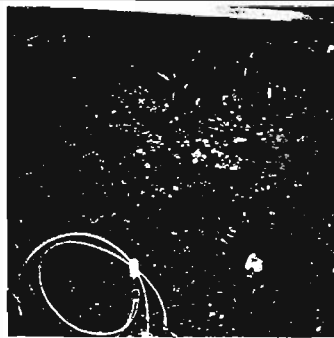
Modas del sector forestal

Rodal
secundario de
piño
modelo 1950

Rodal
secundario de
picea
modelo 1990

Mühleggerköpfl-Tirol del Norte - sitio montañoso / calize

picea / haya

soil solution study with minimal site disturbance

El presentador

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Soil organic matter processes: characterization by ^{13}C NMR and ^{14}C measurements

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Abstract

Soil organic matter (SOM) is a central contributor to soil quality as it mediates many of the chemical, physical, and biological processes controlling the capacity of a soil to perform successfully. SOM properties (e.g. C/N ratio, macro-organic matter) have been proposed as diagnostic criteria of overall soil fitness, but their use is hampered by a poor understanding of the basic biochemical principles underlying SOM processes. The objective of this project was to determine the influence of scrub oak (*Quercus dumosa* Nutt.) and Coulter pine (*Pinus coulteri* B. Don) vegetation on decomposition and SOM formation processes in a lysimeter installation constructed in 1936 in the San Gabriel mountains of southern California. Soil samples archived during construction of the installation, and A horizons sampled in 1987, were fractionated according to density and mineral particle size to isolate the water floatable (macro-organic matter), fine silt and clay fractions. Carbon turnover rates were determined on all fractions from AMS ^{14}C measurements. Solid state CPMAS TOSS ^{13}C NMR was used to semiquantitatively characterize the chemical structure of organic matter on fresh litter and soil fractions. For the two soils, there was a progressive decrease in O-alkyl C, and an increase in alkyl and carbonyl C from the litter to the floatable, fine silt and clay fractions. These compositional differences were due to the oxidative degradation of the litter material, with preferential decomposition of the cellulose and hemicellulose entities and selective preservation of recalcitrant waxes and resins. In all soil fractions, turnover rates of carbon were longer for the pine than for the oak lysimeter (up to 10 times longer). Also under pine, there was a gradual increase in turnover rate progressing from the floatable to the clay fraction, and differences in turnover rates among fractions may be explained based on differences in carbon chemistry. In contrast, under oak, rapid carbon turnover for all fractions suggested intense biological activity in this soil. © 2000 Published by Elsevier Science B.V.

Keywords: Carbon; TOSS CPMAS ^{13}C NMR spectroscopy; AMS radiocarbon measurements; Soil fractions; San Dimas experimental forest; Mediterranean-type ecosystems

1. Introduction

Soil organic matter (SOM) mediates many of the chemical, physical, and biological processes controlling the capacity of a soil to perform successfully; as

such, an assessment of SOM properties is critical to defining overall soil quality. In an effort to characterize quality of agricultural soils from Canada, Gregorich et al. (1994) linked three essential functional roles of organic matter (soil structure, nutrient storage, and biological activity) to a minimum set of soil quality indicators (e.g. total soil organic C and N, macro-organic matter, and carbohydrate content). Such a

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framework is a valuable step towards identification of a quality index that correlates with dynamic SOM processes, but it is hampered by major deficiencies. To allow a more effective interpretation of measurable SOM properties as diagnostic criteria of soil quality, there is a need to better understand the fundamental biochemical processes governing the decomposition and formation of organic matter in soils.

The strong association between organic and mineral soil components has long posed a central challenge to the characterization of SOM chemical structure. A traditional approach has been the extraction of organic substances from soil, such as the use of alkaline hydrolysis followed by acidification, to separate the classical humic and fulvic acid fractions. Undesirable features of this method are an incomplete extraction and a potentially extensive alteration of the extracted products (Stevenson, 1994). In that regard, a most propitious advance in SOM research has been the application of cross polarization-magic angle spinning ^{13}C nuclear magnetic resonance (CPMAS ^{13}C NMR) spectroscopy to whole soils, which dismissed the need for chemical extraction. Continuous development in solid-state ^{13}C NMR technology within the last 10–15 years now allows acquisition of well defined spectra, providing detailed chemical characterization of organic matter in soils (e.g. Oades et al., 1987; Baldock et al., 1992; Preston, 1996).

Atmospheric testing of thermonuclear weapons in the 1950s and 1960s produced a spike input of $^{14}CO_2$ to the atmosphere, which entered terrestrial ecosystems through plants and subsequently enriched SOM with 'bomb' ^{14}C (Goh, 1991). This in situ labeling is a powerful tool for determining SOM turnover rates. In particular, the observed incorporation of 'bomb' ^{14}C into the soil has been used to differentiate SOM into a series of pools with different turnover rates, ranging from seasonal to millennial (e.g. O'Brien and Stout, 1978; Scharpenseel et al., 1989; Trumbore, 1993). The use of accelerator mass spectrometry (AMS) now allows ^{14}C measurements to be made on very small sample sizes, about 10 000 times smaller than needed for decay counting (Trumbore, 1996).

The objective of this study was to elucidate the mechanisms of SOM formation and cycling at the San Dimas lysimeter installation by combining CPMAS ^{13}C NMR analysis and C isotopic techniques. The lysimeter installation was originally constructed in

1936 by the US Forest Service to compare the influence of various plant species on water use and hydrological balances (Patric, 1961). Past research at San Dimas has examined vegetation influence on morphological development (Graham and Wood, 1991), aggregate stability (Graham et al., 1995), total soil C and N (Ulery et al., 1995; Quideau et al., 1998), mineralogy (Tice et al., 1996), and base cation weathering (Quideau et al., 1996). Here, we took advantage of the strictly controlled environment at the lysimeter installation to contrast the influence of scrub oak (*Quercus dumosa* Nutt.) and Coulter pine (*Pinus coulteri* B. Don) vegetation on SOM processes. Specifically, we aimed to evaluate how changes in C chemistry during decomposition processes could be related to measured SOM turnover rates.

2. Materials and methods

2.1. Study area

The lysimeter installation is located within the San Dimas Experimental Forest in the San Gabriel Mountains, 56 km northeast of Los Angeles. Climate at the SDEF is Mediterranean, with a mean annual temperature of 14.3°C, and a mean annual precipitation of 678 mm (Dunn et al., 1988). Large (5.3 by 5.3 m horizontally and 2.1 m deep) earthen-walled pits ('lysimeters') were filled in 1936 with soil material derived on site from the weathering of diorite. To insure homogeneity, the soil material was sieved (<19 mm diameter), and mixed before filling (Colman and Hamilton, 1947). Composite samples of each 7.5 cm incremental soil layer were taken upon filling, and archived in labeled glass jars. Bulk density of the fill material was calculated from the mass of soil added to the pits (Patric, 1961). The lysimeters were planted in 1946 with chamise (*Adenostoma fasciculatum* Hook. and Arn.), hoaryleaf ceanothus (*Ceanothus crassifolius* Torr.), scrub oak, and Coulter pine. Each planting included a single earthen-walled lysimeter, several concrete-lined lysimeters, and buffer strips to eliminate edge effects. In 1960, a wildfire burned the chamise and ceanothus stands to the ground. The oak and pine lysimeters were less affected, with only spotty burning of the litter under oak. Except for some grass cover under the pine, the

lysimeters have remained as virtual monocultures until present. In the summer of 1987, soils were described from pits as reported by Graham and Wood (1991). Differences in earthworm activity corresponding with plant species played a major role in differentiating the soils. Intense earthworm activity under oak had produced a 7-cm thick A horizon composed largely of worm casts; earthworms were absent under pine and the A-horizon was only 1 cm thick (Graham and Wood, 1991).

2.2. Soil sampling and analysis

In 1987, soil A horizons, and in 1994, litter layers, were sampled in triplicate at the oak and pine lysimeters (Graham and Wood, 1991; Quideau et al., 1998). Precise locations disturbed by sampling were carefully noted and are on file with the USDA Pacific Southwest Forest and Range Experiment Station, Riverside, CA. Bulk density was determined using cores (Ulery et al., 1995; Quideau et al., 1996). All soil and litter samples were brought back to the laboratory for drying and weighing. Archived original fill materials corresponding to the A horizon sampling depths were also examined.

Litter samples were dried at 65°C, and ground in a Wiley mill. Soil samples were air-dried, sieved to remove coarse fragments (>2 mm), and fractionated by density and size separation. Soil samples (20 g) were ultrasonically dispersed in water using a Model 550 Sonic dismembrator (Fisher, Pittsburgh, PA) with a 1:10 soil-to-water ratio and a sonification time of 15 min at full power. The sand fraction (>50 μm) was separated by wet sieving (Anderson et al., 1981), and the macro-organic matter (i.e. the floatable fraction) was isolated from the sand fraction by floatation in water. The remaining soil sample was further fractionated using a combination of sedimentation techniques (Genrich and Bremner, 1974) into: coarse and medium silt (5–50 μm); fine silt (2–5 μm); and clay (<2 μm). The floatable, sand, and coarse and medium silt fractions were dried to constant weight at 65°C. The fine silt and clay fractions were flocculated with 1 M KCl, dialyzed against water until free of salt, and freeze-dried. All fractions were analyzed for total C and N by dry combustion.

CPMAS ^{13}C NMR spectroscopy was used to characterize the chemical structure of organic matter in litter, floatable, fine silt, and clay fractions of the two soils. Powdered samples were packed in a 7.5-mm diameter cylindrical zirconia rotor with Kel-F caps and spun at 3.2 kHz in a Doty Scientific MAS probe. The CPMAS TOSS (total suppression of side bands) ^{13}C NMR spectra were obtained using a Chemagnetics 200 (Varian NMR, Palo Alto, CA) operating at 74.5 MHz. Preliminary work established that spinning side bands made a significant contribution to the CPMAS spectra, particularly in the aromatic region. While this resulted in a reduction of the signal-to-noise ratio, we decided to run TOSS on all the samples (Hagemeyer et al., 1991). Additional preliminary work showed that the observable ^{13}C resonances in the samples did not change intensity as a function of the contact time for cross polarization. We used the contact time corresponding to maximum intensity for all experiments (contact time: 1.5 ms; pulse delay: 0.2 s). The number of scans for the litter samples was 10 000 and for the floatable fractions about 50 000. For the fine silt and clay fractions, it was necessary to run the experiment for up to 24 h (400 000 scans).

Graphite targets for ^{14}C analysis were prepared by combusting 0.5 to 1 mg carbon at 900°C with cupric oxide wire in evacuated, sealed quartz tubes, and purifying the evolved CO_2 cryogenically in a vacuum line. Graphite targets were then obtained by catalytically reducing CO_2 using the method of Vogel et al. (1987), and were analyzed for ^{14}C by accelerator mass spectrometry (AMS) at Lawrence Livermore National Laboratory, CA. Radiocarbon data were expressed as F, the fraction Modern carbon (where Modern is by definition 1950), and Δ^{14}C, the ‰ deviation from the ^{14}C/^{12}C ratio of an oxalic acid standard corrected for radioactive decay since 1950. Positive Δ^{14}C values reflect incorporation of bomb ^{14}C, while negative values indicate that ^{14}C has had time to undergo radioactive decay. Average precision for the Δ^{14}C values is ± 7‰.

3. Results

3.1. C and N distribution with particle size

Results of C distribution with particle size have been described in detail elsewhere (Quideau et al., 1998), and are summarized here. In 1987, total C

Table 1

Total C and N concentrations in the original fill material, litter sampled in 1994, and A horizons sampled in 1987 under pine and oak[a]

Sample	C (g kg^{-1})	N (g kg^{-1})	C (% total soil)	N (% total soil)	C (g m^{-2})	N (g m^{-2})
Archived fill material (0–7 cm)						
Fine silt	7.3 (0.1)	0.72 (0.02)	22.5 (0.3)	22.5 (0.4)	57	5.5
Clay	8.7 (0.4)	0.98 (0.02)	41.8 (1.9)	41.8 (0.8)	106	11.8
Whole soil[b]	2.3 (0.1)	0.19 (0.01)	100.0	100.0	253	20.9
Pine (0–1 cm)						
Litter	511 (4)	13.8 (0.5)	–	–	518	14.0
Floatables	341 (8)	16.2 (3.3)	32.2 (6.4)	24.3 (5.2)	55	2.6
Fine silt	41 (6)	3.0 (0.2)	18.1 (2.0)	21.0 (1.5)	31	2.3
Clay	38 (2)	3.4 (0.2)	21.3 (3.1)	30.5 (4.0)	37	3.3
Whole soil[b]	16 (3)	1.0 (0.1)	100.0	100.0	172	10.8
Oak (0–7 cm)						
Litter	448 (16)	10.1 (0.6)	–	–	490	10.9
Floatables	406 (23)	11.9 (0.9)	46.0 (4.5)	26.4 (5.0)	963	28.2
Fine silt	62 (3)	4.3 (0.2)	15.8 (1.2)	21.0 (0.7)	331	23.0
Clay	43 (1)	4.2 (0.1)	14.7 (1.1)	28.1 (2.0)	308	30.1
Whole soil[b]	32 (1)	1.7 (0.1)	100.0	100.0	2093	109.5

[a] Standard errors are indicated in parentheses ($n=3–4$).

[b] <2 mm.

concentration of the A horizon under oak was 32 g kg^{-1} (± 1), twice as high as C concentration of the A horizon under pine (Table 1). Concentrations were higher for oak than for pine in all size fractions, but the distribution of C between fractions indicated that the higher C concentration in the soil under oak principally arose from a greater SOM content (g m^{-2}) in the floatable fraction. By comparison, no floatable fraction was recovered from the archived soil samples, and the majority of C in these samples was associated with the clay fraction. The floatable C fraction corresponds to the macro-organic matter mentioned by Gregorich et al. (1994), a pool of undecomposed or only partially decomposed plant residues which may serve as a readily decomposable substrate for soil micro-organisms. In contrast, C recovered with the fine silt and clay mineral fractions typically includes more decomposed organic materials.

As for C, total N concentration in the A horizons was higher under oak than under pine in the unfractionated soil samples, as well as in the fine silt and clay fractions (Table 1). On the other hand, N concentration was higher for the pine in the floatable fraction as well as in the litter material. As a result of these differences, C/N ratios under oak were much greater than under pine for the litter and the floatable fraction (Fig. 1).

The C/N ratios continued to decrease from the floatable to the fine silt and clay fractions, reaching in these smaller size fractions similar values under the two vegetation types.

3.2. CPMAS TOSS ^{13}C NMR analysis

As illustrated in Fig. 2 for the oak litter sample, NMR spectra showed strong resonances at 72 and 105 ppm, characteristics of C-2, C-3, C-5 and C-1 carbons in cellulose and hemicelluloses, with
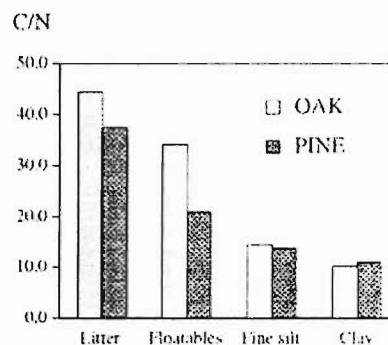

Fig. 1. C/N ratios in litter and particle-size fractions from A-horizon samples under oak and pine.
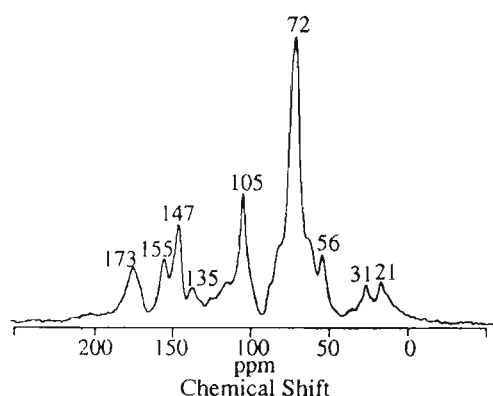
Fig. 2. TOSS CPMAS ^{13}C NMR spectrum of an oak litter sample.

Table 2
Distribution of C species in litter and particle-size fractions from A-horizon samples under pine and oak

Sample	Alkyl C (% total C)	O-alkyl C (% total C)	Aromatic C (% total C)	Carbonyl C (% total C)
Pine				
Litter	10.4	69.0	16.1	4.5
Floatables	21.0	46.6	21.6	10.8
Fine Silt	34.4	33.5	14.6	17.5
Clay	39.4	32.9	9.2	18.4
Oak				
Litter	9.7	62.7	20.1	7.6
Floatables	12.6	58.1	20.1	9.2
Fine Silt	23.9	42.1	18.5	15.4
Clay	32.3	35.1	12.1	20.5

shoulders at 65 and 88 ppm corresponding to the C-6 and C-4 carbons (Duncan, 1987). The peak at 21 ppm was assigned to terminal methyl groups, while the peak at 31 ppm was assigned to alkyl C in long polymethylene type structures (i.e. fatty acids, waxes, resins). The peak at 135 ppm corresponded to C-substituted aromatic carbons, while O-substituted aromatic (phenolic) carbons appeared at 147 and 155 ppm (Hatcher, 1987). Finally, carbonyl functional groups appeared around the peak centered at 173 ppm.

Although TOSS CPMAS NMR is not quantitative for ^{13}C in that the peak intensity does not necessarily reflect the amount of the corresponding carbon, it is semiquantitative if run under carefully controlled conditions at constant speed. Spectra were divided into chemical shift regions as follows (Fig. 2): alkyl C (0–45 ppm); O-alkyl C (45–115 ppm); aromatic C (115–165 ppm); and carbonyl C (165–190 ppm). Magnitude of each chemical shift region was measured by cutting and weighing the corresponding area under the curve, and was expressed as percentage of total area (i.e. relative intensity). The compositional differences noted in progressing from the fresh litter to the floatable, fine silt, and clay fractions were comparable for the oak and pine soils (Table 2). The proportion of O-alkyl carbon progressively decreased from the litter to the floatables, fine silt, and clay, while alkyl and carbonyl C increased. These results are consistent with preferential decomposition of the carbohydrates contained in the litter material, while an increase in the carbonyl region is representative of oxidative degradation processes. The accumulation of alkyl C in the fine silt and clay fractions may arise

either from the selective preservation of recalcitrant litter materials, or from microbial synthesis of new alkyl-type products (Baldock et al., 1992).

While general compositional changes were similar for the two vegetation types, differences were apparent in the magnitude of the changes for the different C species (Table 2). In particular, the decrease in O-alkyl C observed in the floatable and fine silt fractions as compared to the litter was more pronounced under pine than under oak, suggesting a greater decomposition of the cellulose and hemicelluloses contained in the pine litter. This was particularly evident when comparing the litter and floatable fraction. While O-alkyl C made up a greater percentage of total C for the pine litter than was the case for the oak litter, the opposite was true for the floatable fraction. Also, the increase in alkyl C observed in the soil fractions as compared to the litter was greater under pine than under oak.

3.3. SOM turnover rates

The ^{14}C values for the archived and A-horizon samples are presented in Table 3. Negative values for the archived soil fractions reflect the radioactive decay of [14]C. The mean residence time (MRT) for carbon present in the original fill material was estimated using the following equation (Trumbore, 1996):

$$MRT = -\left(\frac{1}{\lambda}\right) \times \ln(F) \qquad (1)$$

where F is the fraction Modern carbon, and λ the rate

Table 3
[14]C content, decomposition constant (k), and mean residence time (MRT) for carbon in particle-size fractions from the original fill material and A-horizon samples under pine and oak

Sample	$\Delta^{14}C$ (‰)	k (per year)	MRT (year)
Archived fill material			
Fine silt	−186.9	0.0005	1660
Clay	−15.0	0.0127	78
Whole soil (<2 mm)	−231.7	0.0040	2130
Pine			
Floatables	220.3	0.0221	45
Fine silt	108.2	0.0149	67
Clay	115.3	0.0128	78
Whole soil (<2 mm)	89.5	0.0126	80
Oak			
Floatables	319.2	0.1155	9
Fine silt	274.2	0.0554	18
Clay	225.4	0.1301	8
Whole soil (<2 mm)	259.4	0.1071	9

constant for the radioactive decay of ^{14}C ($\lambda=0.000121$ per year). The fraction Modern carbon was calculated from the ^{14}C value corrected for the radioactive decay of the oxalate standard since 1950:

$$F = (1.0057 \times 10^{-3} \times ^{14}C) + 1.0057 \tag{2}$$

This yielded a mean MRT of 2130 years for the fill material carbon (Table 3). Carbon associated with the fine silt fraction was relatively old, suggesting no or little recent carbon input to this fraction. The younger age for the clay-sized C, on the other hand, indicated more recent C input, possibly from dissolved organic carbon in percolating waters, or as a result of SOM mineralization processes in larger size fractions.

The 1987 soil samples and associated soil fractions reflected the incorporation of 'bomb' C, as indicated by positive ^{14}C values. A model was developed to simulate carbon fluxes into the soil and decomposition losses. For any given year u, total carbon present in each soil fraction, $C_{t,u}$, was separated into carbon added to the lysimeter soils since planting in 1946, $C_{n,u}$, and carbon still present from the original fill material, $C_{a,u}$:

$$C_{t,u} = C_{n,u} + C_{a,u} \tag{3}$$

with $C_{a,u}$ further defined by

$$C_{a,u} = C_{a,1946} \times e^{-[k_0(u-1946)]} \tag{4}$$

where $C_{a,1946}$ is the amount of carbon originally present in the fill material, and k_0 its decomposition constant, which is equal to 1/MRT as defined in Eq. (1) (Table 3). The increase in C_n with time was fitted to measured carbon contents in the soil using

$$\frac{dC_n}{dt} = b\left[1 - \left(\frac{C_n}{C_{n,1987}}\right)\right]C_n \tag{5}$$

where $b=0.52$ for the pine lysimeter, and 0.275 for the oak lysimeter. Finally, the total ^{14}C content of SOM, $^{14}C_{t,u}$, was obtained using

$$^{14}C_{t,u} \times C_{t,u} = (^{14}C_a \times C_{a,u}) + (^{14}C_{n,u} \times C_{n,u}) \tag{6}$$

where $C_{n,u}$ can be calculated as the following sum for all years over the 1946-u time interval (i.e. x varies from 1947 to year u):

$$^{14}C_{n,u} = \sum I_x \times ^{14}C \times e^{-[(k_n+\lambda)(u-x)]} \tag{7}$$

with I_x equal to the input of carbon to the soil for year x, and ^{14}C equal to average ^{14}C activity of atmospheric CO_2 for the previous year ($x-1$). The decomposition constants corresponding to the carbon added to the lysimeters between 1946 and 1987, k_n, were then obtained by fitting Eq. (7) for the measured ^{14}C values in the different fractions from the 1987 soil samples. Finally, decomposition constants corresponding to the total C present in each soil fraction, k, were calculated as

$$k = \frac{(C_{n,1987} \times k_n) + (C_{a,1987} \times k_0)}{C_{n,1987} + C_{a,1987}} \tag{8}$$

Changes in ^{14}C content with time for total C in the oak and pine lysimeters are plotted in Fig. 3, and calculated decomposition constants for each fraction are presented in Table 3. Under pine vegetation, $^{14}C_t$ increased at a relatively slow rate with time, reflecting dilution of the $^{14}C_n$ signal by the fill material carbon (Fig. 3). The rate of increase was much greater under oak, since proportionally more new carbon was being added to this soil. Also, $^{14}C_t$ for the oak lysimeter showed a decrease after 1974, which may be related to the decrease in ^{14}C concentration for atmospheric CO_2 after 1965. A lag response of 9 years corresponds to the MRT calculated for total C in the soil under oak (Table 3). In contrast, no decrease in ^{14}C was apparent for the soil under pine over the course of the lysimeter history, as total C in this soil was calculated to have a turnover time of 80 years.
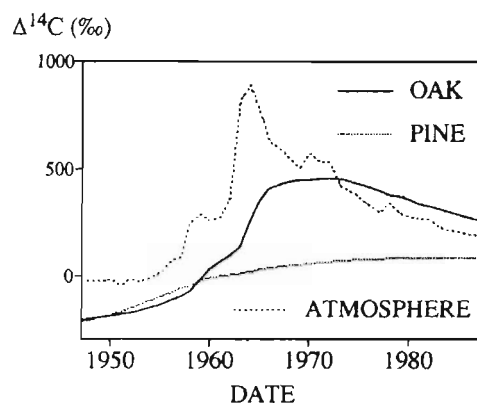
Fig. 3. ^{14}C content in atmospheric CO_2 and in SOM under oak and pine between 1946 and 1987.

4. Discussion

Results from our NMR analyses, such as the observed decrease in O-alkyl from the litter to the macro-organic matter (i.e. the floatable fraction) to the more decomposed organic materials associated with the mineral soil fractions, are in general agreement with changes noted in other studies where ^{13}C NMR was used to characterize decomposition processes (Oades et al., 1987; Preston et al., 1989; Guggenberger et al., 1995). In particular, Baldock et al. (1992) used NMR to analyze SOM from three different orders (Mollisol, Oxisol, and Andosol), and formulated a model describing the chemical changes associated with organic matter decomposition processes. They proposed that the extent of decomposition followed a continuum from the litter to the soil clay fraction. During the initial stage of decomposition, labile structures, such as cellulose and hemicelluloses, are preferentially degraded by the microbial community, resulting in a decrease of the O-alkyl C signal. At that time, there is an increase in the aromatic C signal due to the selective preservation of these more recalcitrant organic compounds (e.g. lignins). Once the O-alkyl C contained in the original litter materials has been degraded, the second stage of decomposition is initiated; aromatic C is decomposed, and a decrease in the aromatic C NMR signal is observed. The third and final stage of decomposition is characterized by the accumulation of alkyl C (Baldock et al., 1992).
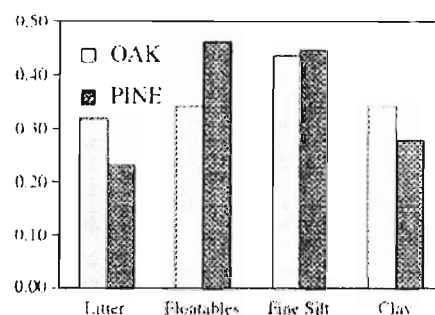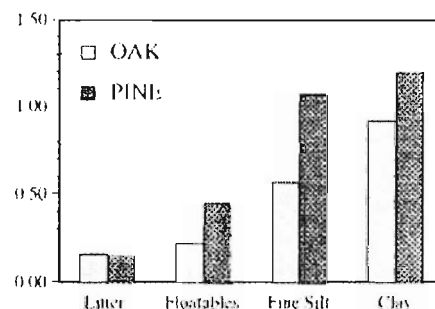

Fig. 4. Aromatic/O-alkyl C and alkyl/O-alkyl C ratios in litter and particle-size fractions from A-horizon samples under oak and pine vegetation.

Based on the Baldock et al. (1992) model, we considered the applicability of using the aromatic/O-alkyl C ratio as an indicator of the first and second stages of decomposition (Fig. 4a). Under both oak and pine vegetation, the floatable fraction exhibited a higher aromatic/O-alkyl C ratio than the corresponding litter, which is consistent with a preferential degradation of O-alkyl C as compared to aromatic C. The increase in the aromatic/O-alkyl C ratio was more pronounced for the pine than for the oak lysimeter. In a similar fashion, the decrease in C/N ratio between litter and floatable fraction was larger under pine than under oak (Fig. 1). Past work at the lysimeter installation reported intense earthworm activity under the oak vegetation, while earthworms were absent under pine (Graham and Wood, 1991). These earthworm species are endogeic; they live within the soil profile but feed on surface organic matter, such as leaf litter, and may contribute to the mixing of fresh plant residues and mineral soil. In contrast, the litter layer

under pine was composed of three distinct horizons: an Oi1 of fresh pine needles, an Oi2 with partially decomposed pine needles, and an Oe of mostly decomposed needles (Graham and Wood, 1991). The C/N ratios of these horizons decreased from 72.5 for Oi1, to 46.6 for Oi2, to 30.3 for Oe, suggesting progressive decomposition with increasing depth within this litter profile. Based on these results, we conclude that the pine litter material had undergone further decomposition than the oak litter at the time of incorporation into the mineral soil.

The fate of plant residues after their incorporation into the mineral soil was followed by examining chemical changes from the floatable to the fine silt and clay fractions (final products of the decomposition processes). Under pine, the aromatic/O-alkyl C ratio exhibited a small decrease in the fine silt fraction as compared to the floatables, and a further decrease in the clay fraction (Fig. 4a). These changes would correspond to the second stage of SOM decomposition as formulated by Baldock et al. (1992). Under oak, the aromatic/O-alkyl C ratio continued to increase between the floatable and fine silt fraction, and only showed a decrease in the clay fraction. These results suggest that the second stage was attained sooner in the decomposition sequence under pine (i.e. in the fine silt fraction), than it was under oak (i.e. in the clay fraction).

In addition to the use of the aromatic/O-alkyl C ratio, the alkyl/O-alkyl C ratio has also been proposed as an index of the extent of organic matter decomposition in forest soils (Baldock and Preston, 1995). For the two lysimeter soils, the alkyl/O-alkyl C ratio progressively increased from the litter to the floatable, fine silt and, finally, clay fraction, which is consistent with an increase in the degree of decomposition (Fig. 4b). While the alkyl/O-alkyl C ratio was similar for the oak and pine litter materials, this ratio was consistently higher under pine than under oak for the soil fractions. Again, this is in agreement with a higher degree of decomposition for SOM present in the soil under pine.

Results from the ^{14}C analyses indicated in all soil fractions longer SOM turnover rates under pine than under oak vegetation (Table 3). Turnover rate of carbon in soils may be influenced by different factors: (1) the chemical structure of the organic fraction; (2) the nature and activity of the soil biota; and (3) the

interaction between organic materials and the mineral matrix, such as the formation of organo-mineral complexes that are typically more resistant to microbial attack than uncomplexed organic materials. Since the lysimeter installation was constructed from homogenized soil material, we can safely assume that differences between the oak and pine SOM turnover rates were due to differences in the organic fraction of the soils and/or in biological activity. For the pine, the MRT of organic C increased from the floatable to the fine silt and clay fraction. This increase in MRT corresponded to an increase in the alkyl/O-alkyl C ratio, and may be explained on the basis of the extent to which plant residues have been processed by the decomposer community. In progressing from the floatable to the clay fraction, there was a gradual increase in the degradation of the original litter material. This, in turns, resulted in a progressive SOM stabilization within the soil matrix, as indicated by longer MRTs in the finer soil fractions. Under pine, the alkyl/O-alkyl C ratio thus appears to be a good index of the degree of SOM decomposition and of its susceptibility to further microbial attack.

In contrast to SOM present under pine, MRTs of soil carbon under oak did not show clear differences between fractions, nor did they correlate well with the alkyl/O-alkyl C ratios (Table 3 and Fig. 4b). While the alkyl/O-alkyl C ratio increased from the fine silt to the clay fraction, estimated MRT was faster for clay-sized SOM than for SOM associated with the fine silt fraction. As plant materials are incorporated into the mineral soil, they either undergo oxidation to CO_2, or they are humified to increasingly resistant organic substances. It appears that intense biological activity in the oak lysimeter (i.e. as indicated by very fast C turnover rates in all soil fractions) resulted in rapid SOM oxidation to CO_2. In turn, this did not allow for the accumulation of SOM with a high degree of humification (i.e. as indicated by NMR results).

Taken together, our results illustrate the importance of combining several experimental approaches, such as NMR and isotopic analyses, to characterize the processes of SOM formation and retention in soils. Under pine vegetation, SOM turnover rates can be explained based on changes in SOM chemistry along the decomposition sequence from the fresh litter to the clay fraction. In contrast, turnover rates of carbon in the soil under oak may be better explained based

on biological activity than on differences in SOM chemistry.

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Trees as carbon sinks and sources in the European Union

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Abstract

The carbon (C) sinks and sources of trees that may be accounted for under Article 3.3 of the Kyoto Protocol during the first commitment period from 2008 to 2012 were estimated for the countries of the European Union (EU) based on existing forest inventory data. Two sets of definitions for the accounted activities, afforestation, reforestation and deforestation, were applied. Applying the definitions by the Food and Agricultural Organization of the United Nations (FAO), the trees were estimated to be a C source in eight and a C sink in seven countries, and in the whole EU a C source of 5.4 Tg year^{-1}. Applying the definitions by the Intergovernmental Panel of Climate Change (IPCC), the trees were estimated to be a C source in three and a C sink in 12 countries, and in the whole EU a C sink of 0.1 Tg year^{-1}. These estimates are small compared with the C sink of trees in all EU forests, 63 Tg year^{-1}, the anthropogenic CO_2 emissions of the EU, 880 Tg C year^{-1}, and the reduction target of the CO_2 emissions, 8%. In individual countries, the estimated C sink of the trees accounted for under Article 3.3 was at largest 8% and the C source 12% compared with the CO_2 emissions. © 2000 Elsevier Science Ltd. All rights reserved.

Keywords: Kyoto Protocol; Article 3.3; Carbon sink; Carbon source; Stock change; Forest; CO_2 emission; Afforestation; Reforestation; Deforestation

1. Introduction

Forests play an important role in the global C cycle. Forest soils have been estimated to contain 790 Pg C, an amount comparable with that in the present atmosphere, and forest vegetation 360 Pg C, an amount comparable with about half of that in the atmosphere and the soils (Dixon et al., 1994; Schimel, 1995). The C in forests originates from the atmosphere, and it continuously cycles between forests and the atmosphere. Thus, changing C stocks in forests can affect the amount of C in the atmosphere. If more C accumulates in forests, they are a sink for atmospheric C. If the C stocks in forests decrease releasing C into the atmosphere, the forests are a source of atmospheric C. The C stocks of forests can change in two ways, on the one hand as a result of changes in forest area and on the other hand as a result of changes in the C stocks on the existing forest area.

The importance of forests for atmospheric CO_2 levels was acknowledged as countries negotiated about their quantified reduction commitments of greenhouse gas emissions in Kyoto in December 1997. According to Article 3.3. of the agreed Kyoto Protocol, some CO_2 sources and sinks of forests "shall be used to meet the commitments" (UNFCCC, 1997a). The sources and sinks to be used were restricted to those "resulting from direct human-induced land-use change and forestry activities, limited to afforestation, reforestation and deforestation since 1990, measured as verifiable changes in C stocks in each commitment period".

The CO_2 sinks and sources of forests in Article 3.3. of the Kyoto Protocol have occasioned discussion and been subject to criticism (e.g. IGBP Terrestrial Carbon Working Group, 1998). The interpretation of the

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Table 1
Applied definitions for afforestation, reforestation and deforestation, and the land area variables of forest inventory data used to represent their areas in this study

Organization	Activity	Definition	Land area variable in forest inventory data used in the calculations
IPCC	Afforestation	"*Forest stands established artificially on lands that have not supported forests for more than 50 years*" (IPCC, 1996)	"*Planting or seeding of non-forest land*" (FAO, 2000)
	Reforestation	"*Planting of forests on lands which have, historically, previously contained forests but which have been converted to some other use*" (IPCC, 1996)	Included in above
	Deforestation	Not defined. FAO definition used	"*Loss to other uses*" (FAO, 1992)
FAO	Afforestation	"*Artificial establishment by planting or seeding of forest on an area of agricultural or other (non-forest) land*" (FAO, 1992)	"*Planting or seeding of non-forest land*" (FAO, 2000)
	Reforestation	"*Artificial or natural re-establishment of forest on previously forested or other wooded land*" (FAO, 1992)	"*Planting or seeding of forest*" plus "*Planting or seeding of other wooded land*" (FAO, 2000)
	Deforestation	"*Change of land use with depletion of crown cover to less than 10%*" (FAO, 1995)	"*Loss to other uses*" (FAO, 1992)

article has not been unanimous. Several views have been presented with respect to the meaning of afforestation, reforestation and deforestation and how the associated C stocks could be accounted for (e.g. Schlamadinger and Marland, 1998; Lund, 1999). The article has been said to provide possibilities to increase cumulative emissions from forests, because it accounts for the C budget of forests only partially and over discontinuous time periods. This would be contradictory to the objectives of the United Nations Framework Convention on Climate Change (UNFCCC) and limit the effectiveness of the whole protocol. It has been claimed that the share of forests and their C sinks and sources the article accounts for are small, although this share and the effect of applying different afforestation, reforestation and deforestation definitions on the C sinks and sources have been quantified only for a few countries (Mäkipää and Tomppo, 1998; Ford-Robertson et al., 1999; Karjalainen et al., 2000; Nabuurs et al., 1999).

To implement the Kyoto Protocol successfully, the consequences of different ways of interpreting Article 3.3. need to be understood. The objectives of the present study are to demonstrate in EU countries (1) how the C sinks and sources of trees that may be accounted for under Article 3.3 of the Kyoto Protocol during the first commitment period from 2008 to 2012 will depend on the applied definition for afforestation, reforestation and deforestation, (2) how these values will differ from the C sink of trees in all forests, and (3) how they will relate to anthropogenic CO_2 emissions.

2. Materials and methods

The definitions for afforestation, reforestation and deforestation we applied to estimate the C sinks and sources of trees that may be accounted for under Article 3.3. of the Kyoto Protocol were those used by IPCC and FAO (Table 1). IPCC and FAO provide different definitions for afforestation and reforestation but only FAO provides a definition for deforestation. We used this deforestation definition also in our IPCC calculation.

The C sinks and sources of trees accounted for under Article 3.3 of the Kyoto Protocol during the first commitment period from 2008 to 2012 applying these definitions were estimated for each EU country. The estimates were based on the latest international forest inventory data (FAO, 1992, 2000). The land area data in the earlier forest inventory report (FAO, 1992) is for the 1980s and those in the more recent one (FAO, 2000) for the early and mid 1990s. The present tree C estimates include C in woody biomass, i.e. wood, bark, branches, twigs, stumps and roots of trees, alive and dead, shrubs and bushes (FAO, 2000).

All forests mean forests on all forest and other wooded land.

The area of "planting or seeding of non-forest land" (FAO, 2000), La (ha year^{-1}), was taken to represent the annual area of human-induced afforestation and reforestation according to IPCC definitions and the annual area of afforestation according to FAO definition (Table 1). The estimate for the annual increase in the C stock of trees on such land between 2008 and 2012, Ba (Mg C year^{-1}), was calculated as

$$Ba = t \times La \times I, \tag{1}$$

where t is the time since 1990 until the middle of the first commitment period (20 years) and I the average net annual increment of tree biomass on forest and other wooded land (Mg C ha^{-1} year^{-1}). $t \times La$ represents the cumulative land area under the activities since 1990 until the middle of the first commitment period. I was taken from the FAO report (2000).

The sum of the areas of "planting or seeding of forest" and "planting or seeding of other wooded land" (FAO, 2000), Lr (ha year^{-1}), was taken to represent the annual area of human-induced reforestation according to FAO definition (Table 1). The estimate for the annual change in the C stock of trees on such land between 2008 and 2012, Br (Mg C year^{-1}), was calculated as

$$Br = t \times Lr \times I - Lr \times S, \tag{2}$$

where t is the time since 1990 until the middle of the first commitment period (20 years), I the average net annual increment of tree biomass on forest and other wooded land (Mg C ha^{-1} year^{-1}) and S the average tree biomass on forest and other wooded land (Mg C ha^{-1}). $t \times Lr$ represents the cumulative reforestation area since 1990 until the middle of the first commitment period. The positive term represents the C sink of trees planted or seeded since 1990 and the negative term the C loss in trees harvested before reforestation during the commitment period. I and S were taken from the FAO report (2000).

The area of "loss to other uses" (FAO, 1992), Ld (ha year^{-1}), was taken to represent the annual area of deforestation (Table 1). This area was not reported in the more recent forest inventory report (FAO, 2000). The estimate for the annual change in the C stock of trees on such land, Bd (Mg C year^{-1}), was calculated as

$$Bd = -Ld \times S, \tag{3}$$

where S is the average tree biomass on forest and other wooded land (Mg C ha^{-1}). S was taken from the FAO report (2000).

The annual change in the C stock of trees accounted for under Article 3.3 of the Kyoto Protocol during the

first commitment period from 2008 to 2012 applying IPCC definitions, B_{IPCC} (Tg C year^{-1}), was calculated as

$$B_{IPCC} = Ba + Bd, \tag{4}$$

and that by applying FAO definitions, B_{FAO} (Tg C year^{-1}), as

$$B_{FAO} = Ba + Br + Bd. \tag{5}$$

These stock changes were compared with the C sink of trees in all forests of the EU in the early 1990s (FAO, 2000) and the anthropogenic CO_2 emissions of the EU in 1995 (UNFCCC, 1997b). The anthropogenic emissions included emissions from energy production, industrial processes, waste and other sources but excluded emissions from land-use change and forestry.

The present estimates of the C sinks and sources of trees accounted for under Article 3.3. of the Kyoto Protocol during the first commitment period from 2008 to 2012 must be considered rough. First, they are predictions of the future, which are uncertain by nature. Second, in calculating the estimates we needed to assume that the annual afforestation, reforestation and deforestation areas remain for the whole calculation period as they were reported in the latest forest inventory reports (FAO 1992, 2000). Any changes in the areas, change the C sinks and sources. Third, we used average growth rates to calculate the C sinks of afforestation and reforestation and average biomasses to calculate the C sources of reforestation and deforestation. Since the afforested and reforested stands to be accounted for will be young, using the average growth rate may overestimate their C sinks. Similarly, the average biomass does not necessarily represent tree
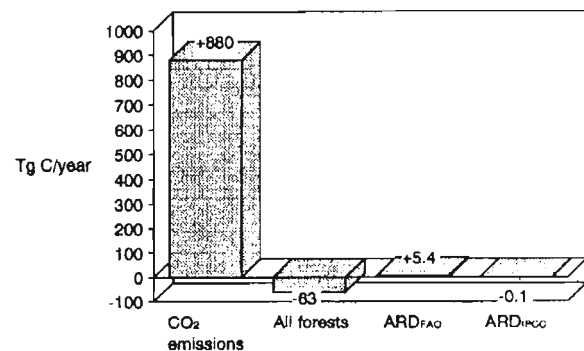

Fig. 1. Annual change in the carbon stock of trees accounted for under Article 3.3 of the Kyoto Protocol in the EU during the first commitment period from 2008 to 2012 applying IPCC (ARD$_{IPCC}$) and FAO definitions (ARD$_{FAO}$) for afforestation, reforestation and deforestation compared with the C sink of trees in all EU forests in the early 1990s (FAO, 2000) and the CO_2 emissions of the EU in 1995 (UNFCCC, 1997b). Sinks for atmospheric C have a negative sign and sources of atmospheric C a positive sign.

Table 2

Annual change in the carbon stock of trees accounted for under Article 3.3 of the Kyoto Protocol in EU countries during the first commitment period from 2008 to 2012 applying IPCC (ARD_{IPCC}) and FAO definitions (ARD_{FAO}) for afforestation (A), reforestation (R) and deforestation (D), annual change in the carbon stock of all trees in the early 1990s (FAO, 2000) and anthropogenic CO_2 emissions in 1995 (UNFCCC, 1997b). The average tree data and the ARD areas used in the calculations are also shown (FAO, 2000, except where else noted). Symbols in italics refer to Eqs. (1) to (5) in the text

Country	Areas			Average tree data		Changes in the C stock of trees							CO_2 emissions in 1995	Changes in the C stock of trees per CO_2 emissions		
	AR_{IPCC} A_{FAO} La 1000 ha yr⁻¹	R_{FAO} Lr 1000 ha yr⁻¹	D_{IPCC} D_{FAO} Ld 1000 ha yr⁻¹ᵃ	Net annual increment I Mg C ha⁻¹ yr⁻¹	Biomass S Mg C ha⁻¹	AR_{IPCC} A_{FAO} Ba Tg C yr⁻¹	R_{FAO} $t \times Lr \times I$ Tg C yr⁻¹	R_{FAO} $-Lr \times S$ Tg C yr⁻¹	D_{IPCC} D_{FAO} Bd Tg C yr⁻¹	ARD_{IPCC} B_{IPCC} Tg C yr⁻¹	ARD_{FAO} B_{FAO} Tg C yr⁻¹	All forests Tg C yr⁻¹	Tg C	ARD_{IPCC} %	ARD_{FAO} %	All forests %
Austria	1.0	7.0	1.0	3.8	147.9	0.08	0.53	−1.04	−0.15	−0.07	−0.58	5.2	16.9	−0.4	−3.4	30.5
Belgium	0.1	5.3	0.0	2.0	58.3	0.00	0.21	−0.31	−0.00	0.00	−0.09	0.2	33.1	0.0	−0.3	0.7
Denmark	1.9	5.8	–	2.0	36.4	0.07	0.23	−0.21	−0.00	0.07	0.09	0.3	16.2	0.5	0.6	1.9
Finland	9.0	118.0	9.5	1.1	28.6	0.19	2.53	−3.38	−0.27	−0.08	−0.93	6.6	15.3	−0.5	−6.1	43.4
France	8.0	30.0	60.0	1.5	49.4	0.24	0.90	−1.42	−2.96	−2.72	−3.25	9.9	105.1	−2.6	−3.1	9.4
Germany	4.2	42.0	0.0	2.8	85.7	0.23	2.31	−3.60	−0.00	0.23	−1.06	14.0	244.0	0.1	−0.4	5.7
Greece	1.3ᵃ	4.0	0.7	0.2	8.0	0.01	0.02	−0.03	−0.01	0.00	−0.02	0.6	24.7	0.0	−0.1	2.4
Ireland	17.0	4.0	–	1.8	14.7	0.60	0.14	−0.06	−0.00	0.60	0.68	0.3	9.3	6.5	7.3	3.8
Italy	9.1ᵃ	12.0	0.0	0.9	37.7	0.17	0.23	−0.45	−0.00	0.17	−0.05	6.9	119.3	0.1	0.0	5.8
Luxembourg	0.1ᵃ	0.0	–	2.4	74.3	0.00	0.00	−0.00	−0.00	0.00	0.00	0.1	2.6	0.1	0.1	3.3
Netherlands	1.2	0.9	0.2	3.9	86.4	0.09	0.07	−0.08	−0.02	0.08	0.07	0.4	50.0	0.2	0.1	0.8
Portugal	28.0	13.0	–	1.7	27.7	0.95	0.44	−0.33	−0.00	0.95	1.06	1.5	13.9	6.8	7.6	10.5
Spain	4.4ᵃ	40.0	4.0	0.4	7.2	0.03	0.29	−0.29	−0.03	0.00	0.01	4.5	63.1	0.0	0.0	7.1
Sweden	2.0	160.0	0.0	1.1	34.6	0.04	3.55	−5.53	−0.00	0.04	−1.94	10.9	15.8	0.3	−12.2	68.5
United Kingdom	22.8	14.4	0.4	1.9	53.0	0.85	0.54	−0.76	−0.02	0.83	0.60	1.8	148.2	0.6	0.4	1.2
Total	110.1	454.2	75.8	1.8	50.0	3.56	11.98	−17.49	−3.45	0.11	−5.39	63.2	877.4	0.0	−0.6	7.2

ᵃ From FAO (1992).

biomass before reforestation or deforestation. Despite these limitations, we think the estimates are appropriate for accomplishing the objectives of this study.

3. Results

In the EU as a whole during the first commitment period from 2008 to 2012, trees accounted for under Article 3.3 of the Kyoto Protocol applying FAO definitions were estimated to be a source for C of about 5.4 Tg year^{-1} (Fig. 1). This was because the C losses on reforestation and deforestation areas, 17.5 and 3.5 Tg year^{-1} respectively, were larger than the C sinks on afforestation and reforestation areas, 3.6 and 12.0 Tg year^{-1} respectively (Table 2). Applying IPCC definitions, the trees were estimated to be a C sink of 0.1 Tg year^{-1} (Fig. 1). The balance was a result of the C losses on deforestation areas, 3.5 Tg year^{-1}, being nearly as large as the C gains on afforestation and reforestation areas, 3.6 Tg year^{-1} (Table 2).

For both these scenarios, the changes in the C stocks of trees accounted for under Article 3.3 of the Kyoto Protocol were very different from the actual change in the C stock of trees in all forests. Accounting for all forests, tree growth exceeded harvesting in each country so that the C stock of trees in the EU increased as much as 63 Tg year^{-1} (Fig. 1, Table 2).

In comparison with the anthropogenic CO_2 emissions of the EU, 880 Tg C in 1995, the C sink of trees in all forests equalled about 7% of the emissions, while that accounted for under Article 3.3 of the Kyoto Protocol applying IPCC definitions was only 0.02% (Table 2). The C source of the trees applying FAO definitions was 0.6% compared with the emissions.

Applying FAO definitions, the trees accounted for under Article 3.3 of the Kyoto Protocol were estimated to be a C source in eight and a C sink in seven countries (Table 2). The fact that the trees constituted a C source was mostly an effect of reforestation: the C loss in trees harvested before reforestation during the commitment period appeared to be larger than the C gain in regrowing trees since 1990. The estimated C source of reforestation was large in countries where the average tree biomass per forest area was large relative to the average growth rate of trees. Such countries were Austria, Belgium, Finland, Germany, Italy and Sweden. In France, the C source was due to a large reported deforestation area. The trees being a C sink was, in turn, mainly an effect of afforestation. The estimated C sink of afforestation was larger than the C source of reforestation in countries where the average growth rate of trees was high relative to the average tree biomass per forest area. This was the case in Denmark, Greece, Ireland, Luxembourg, the Netherlands,

Portugal, Spain and the United Kingdom. Compared with anthropogenic CO_2 emissions, the net C source of the trees was largest, 3–12% of the emissions, in Sweden, Finland, Austria and France. The net C sink was clearly largest compared with the emissions in Portugal and Ireland where it was about 7% of the emissions.

Applying IPCC definitions, the trees accounted for under Article 3.3 of the Kyoto Protocol were estimated to cover a substantial proportion of the C sink of trees in all forests in countries where the C sink of trees in all forests was relatively small (Table 2). In Denmark they covered 25%, in the Netherlands 19%, in Portugal 65% and in the United Kingdom 46%. In Ireland, the C sink of trees accounted for under Article 3.3 of the Kyoto Protocol was even estimated to be 72% larger than the C sink of trees in all forests in early 1990s. On the other hand, in countries where the C sink of trees in all forests was large, the trees accounted for under Article 3.3 of the Kyoto Protocol were estimated to be only a negligible C sink or source. In Sweden, the C sink of the trees accounted for under Article 3.3 of the Kyoto Protocol was 0.4% of the C sink of trees in all forests, in Germany this proportion was 1.6%. In Austria, Finland and France, the trees accounted for under Article 3.3 of the Kyoto Protocol were estimated to be a C source. Compared with anthropogenic CO_2 emissions, the C sink of trees accounted for under Article 3.3 of the Kyoto Protocol was clearly largest, 7% of the emissions, in Ireland and Portugal (Table 2).

4. Discussion

In the EU as a whole, the changes in the C stock of trees accounted for under Article 3.3 of the Kyoto Protocol applying both studied definitions for afforestation, reforestation and deforestation were negligible compared with the C sink of trees in all forests and anthropogenic CO_2 emissions. In contrast, the C sink of trees in all forests was as much as 7% compared with the emissions. This is about as much as the emission reduction target of the EU in the Kyoto Protocol, 8% of the emissions in 1990 (UNFCCC, 1997a). Thus, the C sink of trees in all forests may be considered as important for the atmospheric CO_2 concentration as the targeted emission reduction while the C budget of trees accounted for under Article 3.3 of the Kyoto Protocol with past afforestation, reforestation and deforestation rates seems unimportant.

The C sinks and sources of trees accounted for under Article 3.3 of the Kyoto Protocol may still be important for the C budget of individual countries. The C sink applying IPCC definitions for afforestation, reforestation and deforestation can be large in countries where the present forest area is relatively

small, leaving substantial areas available for afforestation and reforestation. For example, in Ireland and Portugal, the C sink of trees accounted for under Article 3.3. of the Kyoto Protocol was estimated to be equal to 7% of the anthropogenic CO_2 emissions. Interpreting afforestation, reforestation and deforestation according to FAO definitions did not change the result for these two countries. For some other countries it did. Trees accounted for under Article 3.3 of the Kyoto Protocol applying FAO definitions were estimated to be a C source comparable with 3–12% of anthropogenic CO_2 emissions in Austria, Finland, France and Sweden. This would, accordingly, increase the need for reducing CO_2 emissions in these countries to meet the emission reduction targets.

The C budget of trees accounted for under Article 3.3. of the Kyoto Protocol was related to the C balance of trees in all forests only very weakly. If anything, the correlation tended to be negative. In countries where the C sink of trees in all forests was large, in Austria, Finland, France, Germany, Italy and Sweden, the C sink of trees accounted for under Article 3.3. of the Kyoto Protocol was negligible or the trees were even a C source. Thus, Article 3.3 seems irrelevant for the management of forests in these countries, which account for 85% of the actual C sink of trees in all forests of the EU.

The C sinks and sources of trees on afforestation, reforestation and deforestation areas may play a different role in the C budget elsewhere in the world than in the EU. Afforestation of marginal pastures has been estimated to make a significant contribution to meeting the national obligations of the Kyoto Protocol in New Zealand (Ford-Robertson et al., 1999). Considering the estimated huge C emissions from deforestation in the tropics, 1.6 Pg C year^{-1} (Dixon et al., 1994), stopping deforestation and increasing afforestation are certainly important on a world scale.

The present figures of the C sinks and sources of trees accounted for under Article 3.3. of the Kyoto Protocol during the first commitment period from 2008 to 2012 in the EU must be considered as rough estimates, as already discussed in Section 2 of this paper. Despite these limitations, we think they demonstrate how the C sinks and sources of trees accounted for under Article 3.3. of the Kyoto Protocol depend on the applied definition for afforestation, reforestation and deforestation, differ from the C sink of trees in all forests and relate to anthropogenic CO_2 emissions in the whole EU and individual EU countries.

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The nutrient cycling model: lessons learned

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Abstract

The nutrient cycling model (NuCM) is a stand level model that depicts the cycling of N, P, K, Ca, Mg, and S on daily, weekly or monthly time scales. NuCM has been applied to several forest ecosystems (ponderosa pine, red spruce, beech, eastern deciduous, loblolly pine, slash pine, Scots pine, and Norway spruce) to simulate the effects of changing atmospheric deposition, harvesting, species change, precipitation quantity, increased temperature, elevated CO_2, and liming. In some cases (e.g., harvesting, liming), the model output has matched field data quite well; however, it cannot be known whether the model does so because it accurately portrays nutrient cycling processes or simply because of chance. In other cases, NuCM simulations have either failed to match field data (as in the case of the observed chromatographic response of soil solution cations to a nitrate pulse in a beech forest) or produced results that are counterintuitive but as yet untested (as in the case where increased N translocation caused increased leaching). In that the primary purpose of these simulations has been heuristic rather than predictive, the simulation outputs that are either inconsistent with field data or counter-intuitive are of greatest interest. This review of NuCM applications led to the conclusion that the model has been more successful in matching decadal-scale changes in nutrient pools and soils and less successful in capturing intra-annual variations in soil solution chemistry. The NuCM model, like all models, can use improvements and these have been suggested; however, the model as it is has provided valuable insights into nutrient cycling in forest ecosystems, including the potential for short-term soil change and the great importance of nutrient translocation in N cycling. © 2000 Elsevier Science B.V. All rights reserved.

Keywords: Simulation; Harvesting; Soil solution; Deposition; Fertilization; Precipitation; Temperature; N; P; K; Ca; Mg; S

1. Introduction

The need for understanding and predicting the effects of various perturbations (air pollution, harvesting, fertilization, climate change) on forests coupled with the great improvements in computing power and budget cuts in recent years have led to increased use of simulation modeling. The need for modeling seems obvious, given the lack of long-term data sets or other alternatives with which to evaluate these perturbation effects, and the wherewithal for the construction and utilization of models is clearly available. Arguments are often made that simulation models can be used to gain insight into ecosystem functioning (Kimmins and Scoullar, 1979; Aber et al., 1982; Orestes et al., 1994; Johnson et al., 1995a,b; Rastetter, 1996). Rastetter (1996) also notes that some questions are "trans-scientific", meaning that they cannot be answered by science and in fact transcend science. Global change questions almost always fall into this category. In these cases, modeling is the only way to address these questions. But if model portrayals are inaccurate or incomplete, how much insight is really gained by modeling? What gain is there in observing the

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behavior of a system that does not exist in nature? These are serious questions, which are seldom addressed by modelers. All too often, modelers show only the best of their results and discard failures.

The objective of this paper is to review the performance of the nutrient cycling model (NuCM) in a variety of forest ecosystems under a variety of manipulations. In particular, we review the "successes" (defined as cases where model output mimics field data), "failures" (defined as cases where model output fails to mimic field data), and insights gained by using the NuCM. This model, developed in 1988 as part of the Integrated Forest Study (Liu et al., 1991), has been applied to a variety of forest ecosystems and perturbations with varying degrees of success. We hope that this overview of NuCMs performance will help guide its use in the future and perhaps identify some areas for basic research in forest nutrient cycling.

2. The NuCM model

The NuCM model was designed by a team of investigators in the Integrated Forest Study (see Johnson and Lindberg, 1991) and the code was written by Tetra-Tech (Liu et al., 1991). NuCM depicts the cycling of N, P, K, Ca, and Mg at a stand level but also includes the fluxes of major cations (H^+, NH_4^+, Ca^{2+}, Mg^{2+}, K^+, Na^+) anions (NO_3^-, SO_4^{2-}, orthophosphate, Cl^-, HCO_3^-, organic anion) and Si in precipitation, throughfall, and soil solution. Because NuCM was designed primarily for simulating the effects of atmospheric deposition on nutrient cycling processes, its construction emphasizes soil and soil solution chemistry (Liu et al., 1991).

2.1. Nutrient pools

In NuCM, the ecosystem is represented as a series of vegetation and soil components. The overstory consists of one generic conifer and one generic deciduous species of specified biomass and nutrient concentration (foliage, branch, bole, roots). For mixed species stands, average values for biomass and nutrient concentration by component must be used. NuCM also allows an understory which can be divided into canopy, bole, and roots. Maximum potential growth in the model is defined by the user and is constrained in

the model by the availability of nutrients and moisture. The forest floor is simulated from litterfall inputs and litter decay. Litterfall mass inputs are defined by the user, and litter decay is represented as a four stage process, where (1) litter decays to fine litter, (2) fine litter decays to humus and cations, (3) humus decays to organic acids, NH_4^+, SO_4^{2-}, H^+, and CO_2, and (4) organic acids decay to NH_4^+, SO_4^{2-}, H^+, and CO_2. Each stage is a first order equation of the form

$$\frac{-dC}{dt} = -k\theta_t^{(T-20)}C \tag{1}$$

where C is the component in question, t the time, k the reaction rate coefficient (user-defined), θ_t the temperature correction factor, and T the soil temperature (°C).

The soil includes multiple layers (up to 10), and each layer can have different physical and chemical characteristics. The user defines bulk density, cation exchange capacity, exchangeable cations, adsorbed phosphate and sulfate, and four soil minerals and their composition. These inputs define the initial soil exchangeable/adsorbed pools and total pools. Initial total soil N pools are simulated from litterfall and decay, as described above, and user-defined C/N ratios. Vegetation, litter, and soil pools change over a simulation in response to growth, litterfall and decomposition, and nutrient fluxes via deposition, leaching and weathering, as described below.

2.2. Soil and plant processes

The processes which govern interactions among these pools include translocation, uptake, foliar exudation and leaching, organic matter decay, nitrification, anion adsorption, cation exchange and mineral weathering. Translocation, defined as the removal of nutrients from foliage prior to litterfall, is user-specified (as a percentage of foliage nutrient content). Although menus are available for prescribing translocation for both coniferous and deciduous species, translocation operates only on the deciduous species at present because of a coding error (Kvindesland, 1997). Maximum uptake is calculated from biomass and nutrient concentrations; actual uptake is equal to this maximum value when sufficient nutrients are available and reduced when nutrients become limiting. Reduced uptake first allows reduced nutrient

concentrations in plant tissues (by a user-specified percentage) then causes a reduction in growth. Foliar exudation and leaching rates are simulated in the model as proportional to foliar concentrations and user-defined coefficients.

Nitrification is represented in the form of a Michaelis–Menton rate expression:

$$u = \alpha v \left(\frac{[NH_4^+]}{K_s} - [NH_4^+] \right) \theta_N^{(T-20)} \qquad (2)$$

where u is the nitrification rate, α the pH dependency factor, v the maximum nitrification rate at 20°C, K_s the half saturation constant, and $\theta_N^{(T-20)}$ is the temperature-dependency factor where T the soil temperature in °C. Mineral weathering reactions are described in the model using rate expressions with dependencies on the mass of mineral present and solution-phase hydrogen-ion concentration taken to a fractional power. Cation exchange is represented by the Gapon equation

$$\frac{[EC^{a+}](C^{b+})^{1/b}}{[EC^{b+}](C^{a+})^{1/a}} = K_{gp} \qquad (3)$$

where E is the exchange phase equivalent fraction, C^{a+} and C^{b+} the cations of valencies a and b, respectively, K_{gp} the selectivity coefficient, (C^{a+}) and (C^{b+}) the soil solution activity. The model simulates the noncompetitive adsorption of sulfate, phosphate, and organic acid. Phosphate adsorption in the model is represented by a linear isotherm. Sulfate adsorption can be simulated using either linear or Langmuir adsorption isotherms; the Langmuir isotherm was used in these simulations. Unlike most models of its kind (Prenzel, 1994), NuCM simulates pH-dependent SO_4^{2-} adsorption.

2.3. Nutrient fluxes

Precipitation in the NuCM model is defined by input meterological files (typically 1–5 years long) which are repeated in order to generate long-term simulations. Precipitation is routed through the canopy and soil layers and evapotranspiration, deep seepage, and lateral flow are simulated. Potential evapotranspiration (E_{tp}) is calculated as

$$E_{tp} = \frac{F_{et}}{n} (T_m C_e H_c) \qquad (4)$$

where F_{et} is the evapotranspiration factor, which is a function of latitude (r) (Hargreaves, 1974); $F_{et}=2.322-0.0115r$, $r=35°$; $F_{et}=3.434-0.0434r$, $r=35°$; T_m the mean ambient daily temperature (°F), H_c the humidity correction factor (no units). The movement of water through the system is simulated using the continuity equation, Darcy equation for permeable media flow, and Manning equation for free surface flow. Percolation occurs between layers as a function of layer permeabilities and differences in moisture content. Nutrient pools associated with soil solution, the ion exchange complex, minerals, and soil organic matter are all tracked explicitly by NuCM. Wet deposition is calculated from precipitation amounts and user-input air quality files which define precipitation concentrations on a monthly basis. Dry deposition is calculated from air concentrations in the air quality files combined with user-defined deposition velocities and simulated leaf areas. A menu allows the user to reduce or increase deposition amounts by specific multipliers for the purpose of running deposition scenarios. Leaching is calculated from soil water flux and simulated soil solution concentrations using the soil chemical and biological algorithms defined above for each soil horizon.

2.4. Model output

NuCM output can be viewed on user-selected screens displaying biomass, solution concentrations, soil concentrations, water fluxes, and nutrient pools and fluxes. Output files (ASCII) can also be produced which contain values displayed on the screens. NuCM provides output of solution chemistry on a daily, weekly, or monthly basis, according to user preference. Simulated biomass and soil exchangeable/adsorbed nutrients are provided on a monthly basis, and simulated nutrient pools and fluxes are provided on an yearly basis.

3. Applications of NuCM

NuCM has been applied to a variety of sites and manipulations since its inception with varying degrees of "success". The model can be applied to time frames between 1 and 100 years; typically, the simulations were run for 20–50 years. NuCM has been

Table 1
Sites calibrated for NuCM

Site	Species	Age (years)	Soils and parent material	MAT (°C)[a]	MAP (cm)[b]
Smoky Mountains, NC, USA	*Picea rubens*	300+	Umbric Dystrochrepts (sandstone)	6.0	151
	Fagus grandifolia	>80	Umbric Dystrochrepts (shale)	6.0	
Coweeta, NC, USA	Mixed deciduous	1–80	Typic Hapludult (schist)	12.5	138
	Pinus strobus	30	Typic Hapludult (schist)	12.5	138
Walker Branch, TN, USA	Mixed deciduous	1–80	Typic Hapludult (dolomite)	14.2	136
Duke, NC, USA	*Pinus taeda*	23	Typic Hapludult (igneous rock)	14.5	113
Bradford, FL, USA	*Pinus elliotii*	24	Haplaquods (marine sands)	21.0	112
Barton Flats, CA, USA	*Pinus ponderosa*	40+	Xerumbrepts (granite)	11.0	54
Little Valley, NV, USA	*Pinus jeffreyii*	110	Aquic cryumbrept (granite)	5.0	70
Nordmoen, Norway	*Picea abies*	30–40	Typic udipsamment (glacifluvial dep.)	4.3	74
Åmle/Ås, Norway (soil columns)	*Pinus sylvestris*	4	Typic udorthent (alluvial outwash)	5.3	100

[a] Mean annual temperature.

[b] Mean annual precipitation.

applied to northern and high-elevation coniferous forests, temperate coniferous and deciduous forests, semi-arid coniferous forests, and soil columns with coniferous seedlings. A listing and brief description of the sites for which the model has been calibrated is given in Table 1. The interested reader is referred to the referenced papers for more details on each. The following is a brief summary of simulation results that we feel are most representative of the strengths and weaknesses of the model and some of the insights we have gained as a result.

3.1. "Successes": simulation results mimic field data

By "successes", we mean that the model output mimicked the general patterns observed in the field, sometimes even before field observations were made (prediction). Among the "successes" for NuCM, we include: (1) harvesting and species change in mixed deciduous and loblolly pine forests, (2) response to liming in a deciduous forest, (3) prediction of high NO_3^- in ponderosa pine forests, (4) response to artificial acid "rain" and N fertilization in a soil column experiment, and (5) nutrient cycling in a Norway spruce stand.

3.1.1. Harvesting and species change

The NuCM model was used to simulate the effects of atmospheric deposition and harvesting and species change (mixed deciduous and loblolly pine) at Duke,

NC for the purpose of investigating the relative effects of these perturbations on long-term changes in soil nutrient status and growth (Johnson et al., 1995a, 1998). The simulations suggested that species, specifically, loblolly pine versus mixed deciduous, had a much greater effect on soil nutrient status than atmospheric deposition did. Specifically, high rates of Ca uptake and accumulation by deciduous species caused reductions in soil exchangeable Ca^{2+} compared to a loblolly pine stand. This in turn caused reduced Ca^{2+} leaching and greater Mg^{2+} leaching in the deciduous stand compared to loblolly pine stand. The greater Mg^{2+} leaching occurred despite the fact that exchangeable Mg^{2+} was reduced in the deciduous stand, also. This seemingly counterintuitive result occurred because, while both exchangeable Mg^{2+} and Ca^{2+} decreased in the deciduous stand, the Mg^{2+}/Ca^{2+} ratio increased.

There are no data for the Duke site with which to compare these simulations; however, the simulation results are very similar to those obtained in a field study near Oak Ridge, TN (Johnson and Todd, 1987). (Unfortunately, the Walker Branch site at Oak Ridge was not calibrated at the time of this simulation exercise.) In the Oak Ridge study, as in the simulations, high rates of Ca uptake by deciduous species had apparently reduced soil exchangeable Ca^{2+} and Ca^{2+} leaching rates compared to the loblolly pine stand. Also, as in the case of the simulations, Mg^{2+} leaching was greater in the deciduous stand
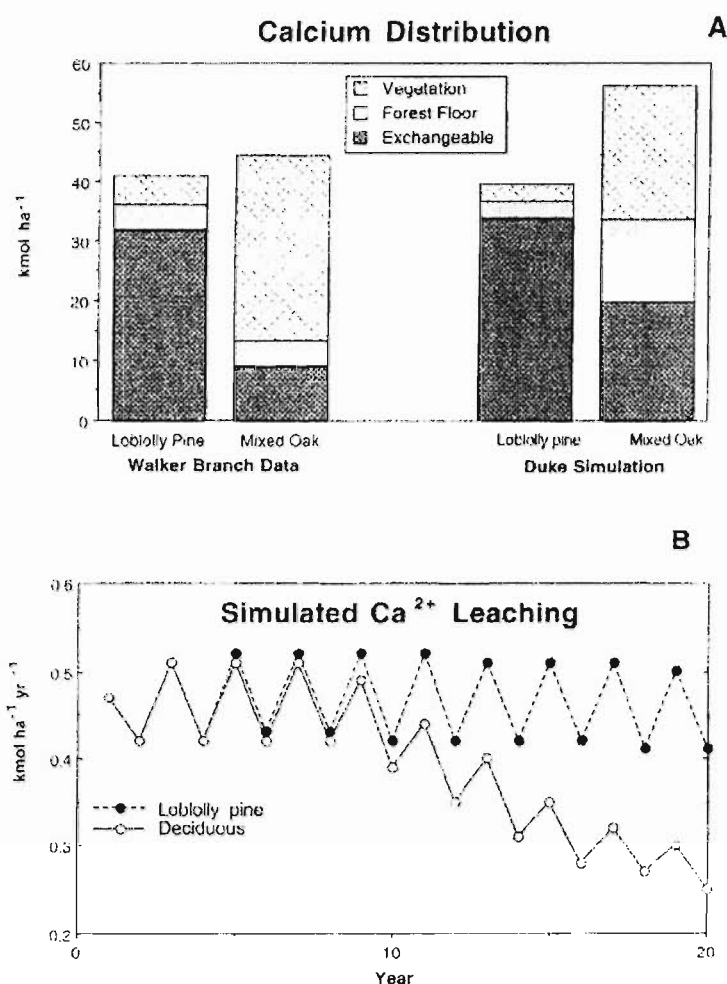
Fig. 1. Ca distribution in loblolly pine and mixed oak stands near Walker Branch Watershed, TN and simulated for the Duke, NC site (from Johnson and Todd, 1987 and Johnson et al., 1995a) (A). Simulated Ca^{2+} leaching at the Duke site (B).

(despite the lower exchangeable Mg^{2+}) because of greater Mg^{2+}/Ca^{2+} ratio. Fig. 1 shows the Ca distributions at the Oak Ridge sites and in the simulations.

Aside from producing Ca and Mg distributions and fluxes similar to those observed elsewhere, the NuCM simulations for Duke site suggested that estimates of nutrient fluxes from only a few years measurement can be misleading because of temporal changes in leaching rates. Both field data and simulation results showed lower Ca^{2+} leaching beneath the mixed oak than beneath the loblolly pine stand near the end of the

rotation, but the simulation results suggested that this difference developed only after soils began to change (15–20 years). Thus, using 2–3 years data near the end of the rotation to estimate long-term budgets, as is typically done in field studies (e.g., Johnson and Todd, 1987), can be seriously misleading.

3.1.2. Liming

The NuCM model was calibrated (Johnson et al., 1993) and later used to simulate the long-term effects of liming on Watershed 6 at Coweeta Hydrologic Laboratory, NC with the aim of comparing the
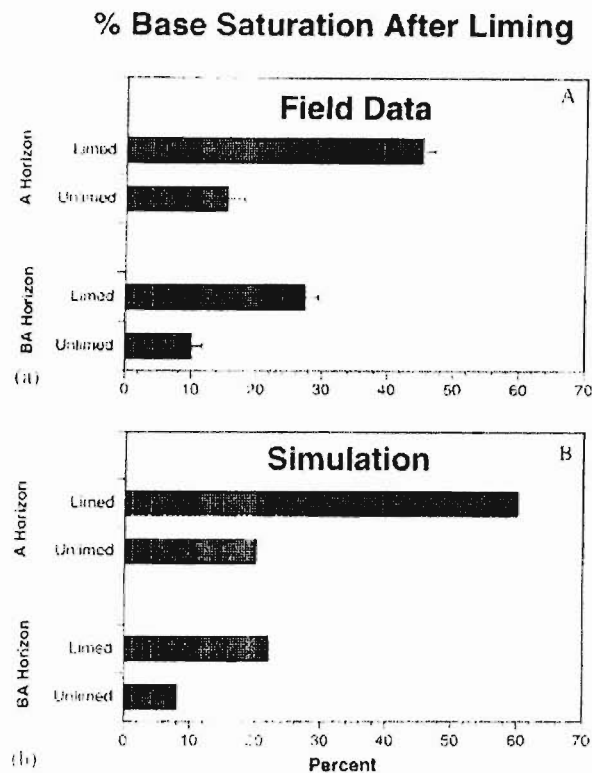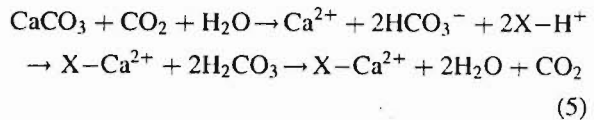
% Base Saturation After Liming

Fig. 2. Measured (A) (Montagnini et al., 1986) and simulated (B) base saturation of 23 years after liming on Watershed 6 at Coweeta, NC (after Johnson et al., 1995b).

simulation output with field results. The results of these simulations can be regarded as a mixed "success". NuCM output for Watershed 6 mimicked the patterns in exchangeable Ca^{2+} and base saturation 23 years after the application of dolomitic lime at Coweeta (Johnson et al., 1995b). The results for base saturation are shown in Fig. 2. The simulations did not, however, mimic field results in the case of Mg^{2+}, where field data showed most retention in surface horizons and the simulations showed most retention in the BC horizon (where most exchangeable Mg^{2+} was originally). In all likelihood, the responses of exchangeable Mg^{2+} could be mimicked if initial conditions in the model calibration were modified.

It is not surprising that NuCM was able to successfully simulate liming. The reactions of lime with soil are relatively straightforward, and NuCM is strong in soil chemistry and cation exchange. Both in the field and in the model, lime is retained in the upper portions of the soil for prolonged periods after liming because of the absence of mobile anions to move the applied Ca^{2+} deeper into the profile. After the limestone dissolves, we have:

$$CaCO_3 + CO_2 + H_2O \rightarrow Ca^{2+} + 2HCO_3^- + 2X-H^+$$
$$\rightarrow X-Ca^{2+} + 2H_2CO_3 \rightarrow X-Ca^{2+} + 2H_2O + CO_2$$
$$(5)$$

where X refers to the exchangeable phase. Thus, the anion (HCO_3^-) entering the soil with the Ca^{2+} is converted to water and CO_2, leaving no means of further transport of the applied Ca^{2+} unless a mineral acid is applied.

3.1.3. Prediction of high soil solution NO_3^- in a ponderosa pine stand

Perhaps one of NuCMs most notable successes was the prediction of high summertime soil solution NO_3^- concentrations in ponderosa pine stands in the San Bernardino Mountains of CA. The model was calibrated for Barton Flats, a moderately polluted area in a 55 km transect from west to east in the San Bernardino Mountains east of Los Angeles with the aim of simulating the effects of varying N deposition. Nitrogen deposition rates ranging from 1.4 to 95.6 kg ha^{-1} per year were simulated using NuCM, covering the range observed in an east to west transect in the region (from 5 to 45 kg ha^{-1} per year; Fenn et al., 1996). Soil and plant responses were as expected, for the most part (Fenn et al., 1996). One surprising result was the prediction of very high soil solution NO_3^- concentrations in late spring–summer in these sites (200–1000 μmol_c l^{-1} for the range of N deposition values observed in the field), even under the lowest N deposition rates. To check these predictions, naturally-wet soils were sampled along the transect and soil water was removed by centrifugation. The results confirmed NuCMs predictions in principle: measured soil solution NO_3^- concentrations ranged from low to very high values (10–1200 μmol_c l^{-1}). The high measured NO_3^- concentrations do not necessarily imply that N leaching rates were high because they tended to occur in the dry season.

3.1.4. Artificial acid rain and N fertilization

NuCM has been applied to a lysimeter experiment where soil columns planted with Scots pine saplings

had been treated with artificial acid "rain" (pH 3) and increased doses of N (30, 90 kg N ha^{-1} per year) (Sogn et al., 1995; Sogn and Abrahamsen, 1997) with the aim of comparing model output with field data. The soil columns were 60 cm deep and soil solution and leachate were collected below the horizons (O, E and Bs) and in the bottom below the BC-horizon. The lysimeters were shielded from natural precipitation by a roof. The application of NuCM to the lysimeter experiment may also be regarded as a mixed success (Sogn et al., 1995; Sogn and Abrahamsen, 1997). By increasing the SO_4^{2-} adsorption capacity relative to the calibration value, the observed level of SO_4^{2-} leaching at higher H_2SO_4 load (pH 3) was successfully simulated. So too was the increased leaching of base cations due to the acid "rain". Close agreement between observed and simulated total Al concentrations was found for the mineral soil horizons (BS and BC), but the model underestimated the Al concentrations in the O horizon. In NuCM, the definition of organic Al is limited to the complex $Al^{3+}R^{3-}$, when in reality Al^{3+} may also complex other fractions (e.g. with partly dissociated organic acids). The formulations connected to organic Al control may be one of the areas where NuCM should be changed in order to incorporate more recent understanding of the organic Al recipitation/dissolution processes.

The major effects of N addition in doses of 90 kg N ha^{-1} per year were also relatively well simulated by the model (Sogn and Abrahamsen, 1997). By increasing the nitrification rate relative to the calibrated value, the observed yearly mean values of NH_4^+ and NO_3^- leaching were approached by NuCM. From calibration to control, however, it was clear that NuCM had problems simulating the seasonal variation of N leaching. The NO_3^- leaching during the dormant season was indeed overestimated by the model. However, when high amounts of N were added the agreement between observed and seasonal variations in N leaching were improved relative to the control. The soil/plant systems response to elevated N input seemed to reduce the usual noise of seasonality, which apparently increased the conformity between observed and simulated N leaching. As for acidification, increased leaching of base cations following increased N leaching was satisfactorily simulated by the model (so too was the increased Al leaching from mineral soil horizons). The Al concentrations in the organic horizon were underestimated by the model.

3.1.5. Nutrient cycling in a Norway spruce stand

The NuCM model was calibrated and tested for a site at Nordmoen, Norway, one of the Integrated Forest Study sites for which the model was designed to be used. A detailed study appraising the ability of NuCM to simulate nutrient cycling in this stand revealed both advantages and disadvantages of the model (Kvindesland, 1997). NuCM successfully simulated the concentrations of SO_4^{2-} and Al in all the soil horizons, and the concentrations of H^+, Ca^{2+}, Mg^{2+} and Na^+ in the O and E horizons. NuCM failed to simulate the organic and inorganic monomeric Al in all soil horizons. The dynamics of soil solution dissolved organic carbon (DOC) concentrations were well simulated by the model, but the average concentration in the simulation was too low. This low DOC level could be due to a lack of process formulations describing root exudation in NuCM, or poor process description of DOC production from litter. The NO_3^- concentrations were poorly simulated, especially during the winter. During the winter period, the model simulated high concentrations of NO_3^- in throughfall. Since the model lacks a description of immobilization of N by soil microbes, this lead to simulations of high NO_3^- concentration in soil water during the winter periods.

Regarding simulation of throughfall concentrations in the Norway spruce stand, NuCM successfully simulated the concentrations of K^+ and Mg^{2+}, while the concentrations of Ca^{2+} and H^+ were poorly reproduced. NuCM describes many canopy processes, but the sub-model is difficult to calibrate when only bulk deposition and throughfall are routinely measured. At the Nordmoen site, a large disappearance of inorganic N and production of organic N are observed in canopy. Such a process is not described in NuCM, and thus the simulated nitrogen concentrations in throughfall were a failure at this site.

Regarding fluxes in the different forest ecosystem compartments, the cycling of NH_4^+ and Mg^{2+} were successfully simulated. The nutrient cycling of Ca^{2+} in particular and K^+, especially exudation, uptake, and litterfall were poorly simulated by NuCM. This is due to a lack of a process describing translocation/accumulation of nutrients in conifers in the model.

3.2. "Failures": simulation results fail to mimic field data

We define "failure" in this context as the failure of the model to mimic the general patterns observed in the field data. The reader should note that "failure" is in quotes, because in many ways these comparisons have much greater value for our understanding of nutrient cycling processes than the "successes" do. There are many cases where NuCM has failed to either mimic or predict field data, even where the model's overall performance was deemed "successful" as noted above. In this section, we will focus on some additional "failures" which we feel have provided insight into the nature of important processes in the field which NuCM inadequately represents.

3.2.1. Chromatographic leaching of cations in a beech site

One of the most illuminating "failures" of NuCM was in failing to reproduce the cation leaching patterns in a beech forest soil in the Great Smoky Mountains Park, NC. In the autumn of 1985, three months after establishing lysimeter plot, a large NO_3^- pulse occurred which caused concurrent pulses of cations in soil solution. The cause(s) of this NO_3^- pulse are not known but also are not particularly relevant for the testing of NuCM in this instance. As the NO_3^- pulse leached through the soil profile, it caused pulses in all cations in soil solution, as theory would predict. As the pulse passed through successively deeper horizons, the peaks in cation concentrations separated, just as in a chromatographic column. Fig. 3 illustrates the patterns for Mg^{2+}, Ca^{2+}, and Al^{3+} observed in the field (Johnson, 1995). NuCM failed to reproduce this chromatographic effect: all cations peaked simultaneously with NO_3^- in all horizons (Fig. 4). This response occurred even when the soil was broken into 10 horizons (as many as the model will allow). The consequences of this failure could be significant in terms of Al toxicity: because of the separation of peaks, Al/Ca ratios in the deepest horizons of the beech site were much higher than they would have been if all cations had peaked simultaneously (Fig. 5, top). NuCM, therefore, greatly underestimated the responses of Al/Ca ratios to the NO_3^- pulse (Fig. 5, bottom).

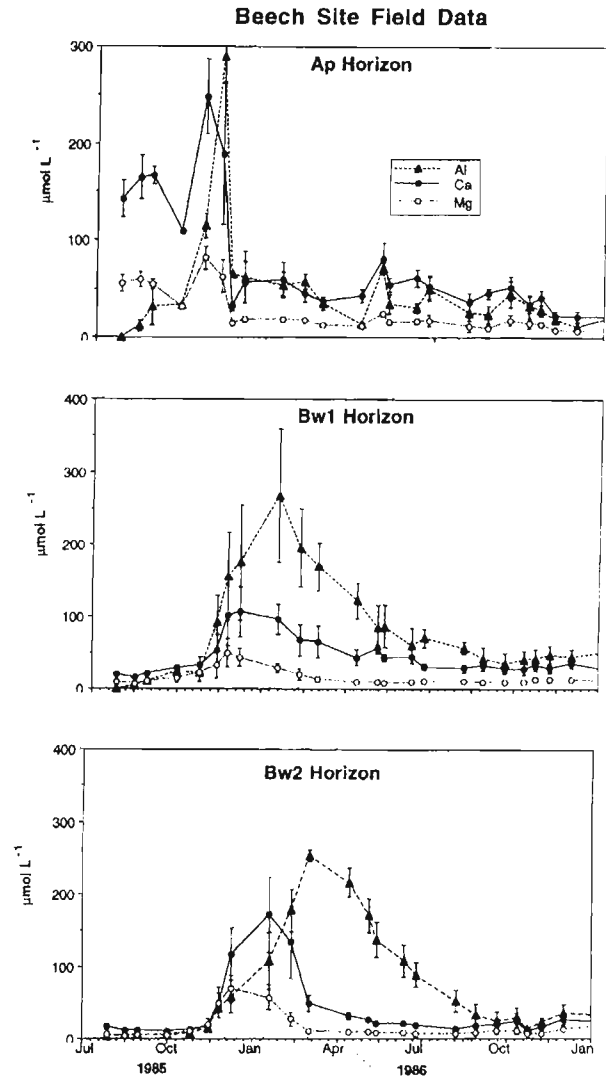These results suggest that the resolution of soil layers in NuCM is too coarse to accurately predict

Fig. 3. Soil solution Al, Ca^{2+}, and Mg^{2+} during a nitrate pulse in a beech site in the Great Smoky Mountains, NC (after Johnson, 1995). (Reprinted by permission of the Soil Science Society of America).

the short-term temporal dynamics of soil solution. A chromatographic model (e.g., Mansell et al., 1988) would probably be more appropriate to stimulate soil solution dynamics on such time scales.

3.2.2. Responses of soil solution to precipitation changes in a deciduous forest

To investigate the potential effects of changing precipitation quantity on a deciduous forest ecosys-
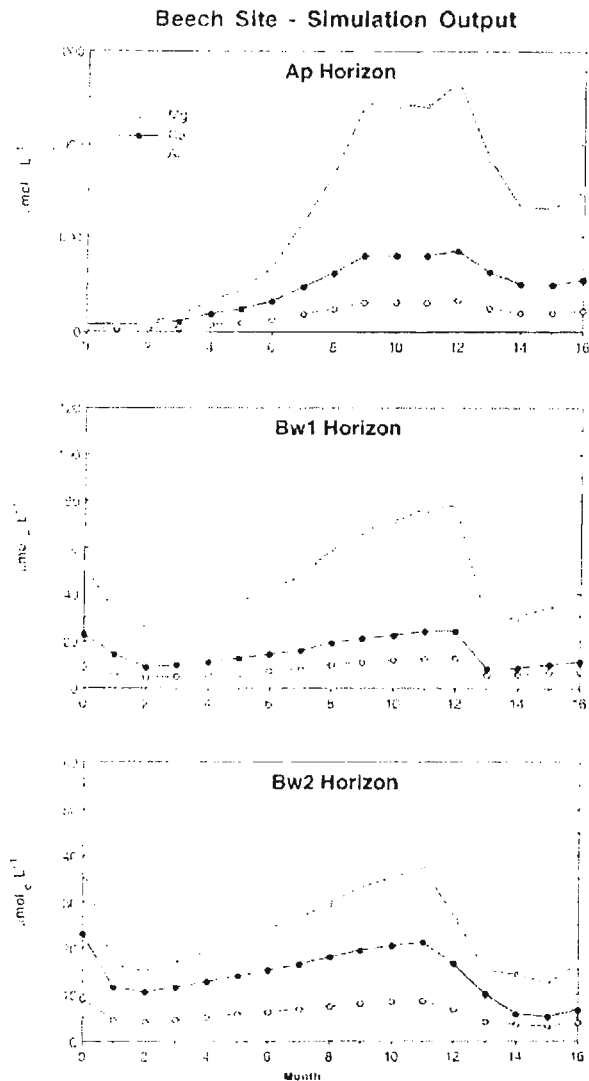
Beech Site - Simulation Output

Ap Horizon

Bw1 Horizon

Bw2 Horizon

Fig. 4. Simulated soil solution Al^{3+}, Ca^{2+}, and Mg^{2+} during a nitrate pulse in a beech site in the Great Smoky Mountains, NC (after Johnson, 1995). (Reprinted by permission of the Soil Science Society of America).

tem, an experiment was established on Walker Branch Watershed, TN which modified the amount of throughfall at −33%, ambient (no change), and −33% using a system of rain gutters and sprinklers (Hanson et al., 1995). NuCM was calibrated for this site, and the effects of the treatments were simulated for the purpose of testing the model and for long-term assessments of treatment effects (Johnson, 1999). The simulations matched the patterns during fourth year of treatment fairly well for soil solution conductivity,

Al/Ca Molar Ratios   A
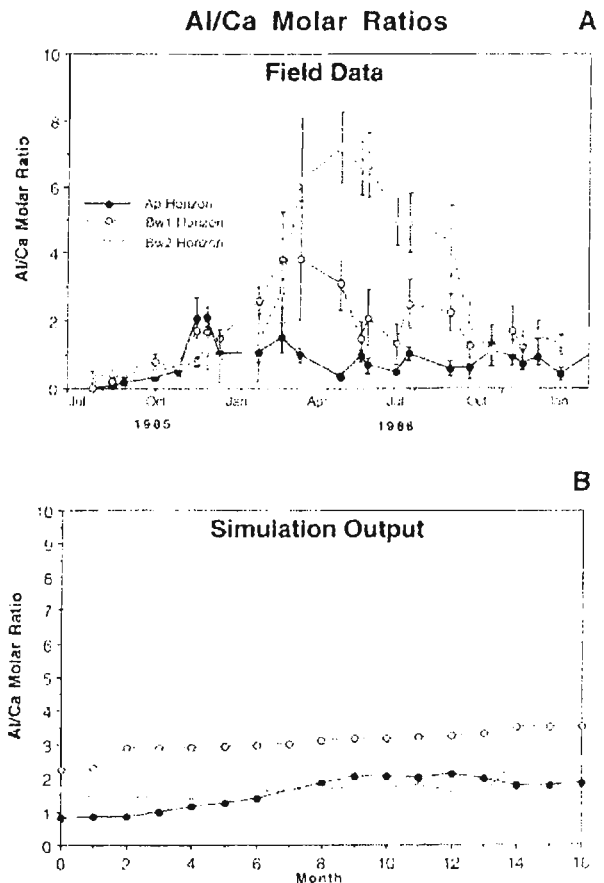
Field Data

Simulation Output

Fig. 5. Soil solution Al/Ca molar ratios during a nitrate pulse in a beech site in the Great Smoky Mountains (A). Simulated soil solution Al/Ca molar ratios during a nitrate pulse in a beech site in the Great Smoky Mountains (B) (after Johnson, 1995). (Reprinted by permission of the Soil Science Society of America).

SO_4^{2-}, Ca^{2+}, Mg^{2+}, and Na^+ but poorly for Cl^- and K^+. The results for Cl^- are shown in Fig. 6. Simulated soil solution Cl^- varied seasonally by a factor of three, whereas there was little seasonal variation in the field data. In the case of K^+, the simulations indicated very little seasonal variation and very minor treatment effects. In the field, there was a large seasonal variation in K^+ in the dry treatment and a very large treatment effect during peak concentrations. In contrast to the field results, the NuCM simulations showed only a very slight responses of SO_4^{2-}/Cl^- ratios to treatments in the fourth year. There were larger but inconsistent responses of SO_4^{2-}/Cl^- ratios to treatments in 1–3 years of the simulation.
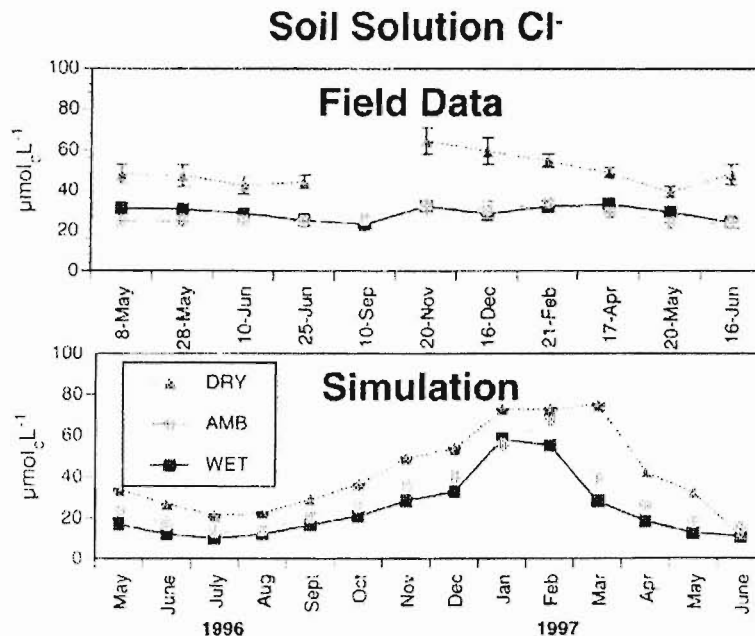
Fig. 6. Actual and simulated soil solution Cl⁻ with −33, +33%, and no change in precipitation at Walker Branch Watershed, TN (after Johnson et al., 1998).

These comparisons of field and simulation results show that NuCM underestimated soil buffering for Cl^- and overestimated buffering for K^+ at the Walker Branch site. In the case of Cl^-, we hypothesized that buffering by adsorption and desorption accounted for the stability of concentrations in the field data. Some soils are known to have Cl^- adsorption properties (Johnson and Cole, 1980), especially when Fe and Al hydroxide concentrations are high, as is the case in the Ultisol at Walker Branch. The model does not currently allow for this Cl^- adsorption. In the case of K^+, we hypothesized that macropore flow accounted for the observed concentration spike in the field data.

3.3. NuCM results with no field data available

Much of the modeling with NuCM has had as its primary purpose the exploration of the collective implications of what we currently know (or think we know) about nutrient cycling processes in forest ecosystems. In short, we wanted to discover if there were any surprises and/or insights to be gained by formalizing and exercising our knowledge of nutrient cycling processes into a computer code. As antici-

pated, there have been many surprises and some insights, mostly as a result of the "failures" such as those described above. Below we illustrate some of the surprises and insights gained which have yet to be tested.

3.3.1. Soil change

In the early days of assessing acid deposition effects on soils, many predicted that soils were very inert and slow to change because simple budget calculations indicated that pool sizes were large compared to fluxes into and out of the soil profile (McFee, 1980; Reuss and Johnson, 1986). Reuss and Johnson (1986) used the terms "capacity" and "intensity" to refer to changes in the soil exchangeable and soil solution phases, respectively. In a hypothetical, typical example of soil total exchangeable pool sizes and fluxes, the authors concluded that, "the total exchangeable acidity is far in excess of annual acid inputs. Therefore, substantial changes in total exchangeable acidity would not be expected unless the system were dramatically affected for many decades" (Reuss and Johnson, 1986, p. 8). Since that statement was written, several field studies have shown declines in soil

exchangeable bases over periods of decades, even in some sites with relatively low atmospheric inputs (Johnson et al., 1988, 1991; Binkley et al., 1989; Falkenggren-Grerup and Eriksson, 1990; Falkenggren-Grerup and Tyler, 1992; Johnson et al., 1994; Knoepp and Swank, 1994; Bergkvist and Folkeson, 1995). Vegetation uptake of base cations has proven to be a major factor in causing soil change in many cases (Johnson et al., 1988; Johnson and Todd, 1990; Johnson et al., 1994; Richter et al., 1994). While the conclusions of Reuss and Johnson (1986) remain valid for total soil exchangeable pools (e.g., soil content to a depth of 0.5–2 m), they need revision when considering changes in exchangeable cations among various soil horizons. If fluxes into and out of most soils are compared with soil nutrient contents, then the possibilities for change over periods less than centuries seem very limited. However, if soils are divided into their natural horizons and fluxes into and out of these horizons are known from lysimetry, then the possibility of change over much shorter time periods becomes quite obvious. And, of course, if the soil is viewed as a continuous chromatographic column, the possibilities of local change are almost instantaneous. In our modeling analyses of extremely acidic red spruce forest soils of the southern Appalachians, for example, we concluded that "the interactions among soils and solutions are so intimately and immediately intertwined as to suggest that differentiating between intensity and capacity type effects may not be useful and is perhaps misleading" (Johnson et al., 1996, p. 1306).

Thus, field data and rethinking of assumptions have changed the views of many soil scientists regarding the possibilities for soil change. Despite its failure to reproduce the chromatographic responses in the beech soil described above, NuCM has added insights which are yet to be tested into the ways in which soils might change. NuCM nearly always shows seasonal fluctuations in soil exchangeable base cations, as has been observed in the field in some sites (Haines and Cleveland, 1981; Johnson et al., 1988). NuCM also suggests that long-term trends in soil change may not be unidirectional: simulated exchangeable bases often show both increases and decreases over certain periods of time, reflecting adjustments in the overall cycling of these nutrients over time. This frequency of change may not visible on the decadal time scale of

most soil sampling protocols. NuCM has also suggested inter horizontal interactions in response to acidification pressure that are counterintuitive at first glance yet reasonable upon reflection and certainly mathematically possible.

A typical example of such behavior for the Coweeta simulations in response to reduced S deposition is shown in Fig. 7 (from Johnson et al., 1998). In the A horizon, there is a continuous downward trend (with seasonal oscillations) in exchangeable Ca^{2+} that is slightly mitigated by reduced S deposition. In the BA horizon, the seasonal oscillations are damped, but there appears to be another longer-term oscillation of greater amplitude and much longer frequency (first an increase then a decrease). In the BC horizon, both seasonal and longer-term oscillations are nearly damped out. However, reduced S deposition has the counterintuitive effect of causing reduced exchangeable Ca^{2+} because of lower inputs from upper horizons.

3.3.2. Effects of nitrogen translocation

In order to simulate the effects of elevated CO_2 on nutrient cycling at the Walker Branch and Duke sites, several scenarios were run where the target growth rate and litterfall were increased, foliar N was decreased, and soil N mineralization were increased (Johnson, 1999). The results of this exercise were interesting but will not be discussed in detail here; the reader is referred to Johnson (1999) for full details. One interesting result that arose from these simulations was the very large effect of translocation on growth and N cycling in the Walker Branch site. As noted previously, the present NuCM code does not allow for translocation in coniferous species, and thus the effects of translocation on simulation results in both the baseline and manipulated scenarios were tested for the deciduous Walker Branch site.

Table 2 shows the results of the simulations at Walker Branch in the base case where translocation was set at 47% (the value obtained from field estimates) and 0%. The effects of translocation on both biomass and N cycling were considerable: biomass was 17% greater and vegetation N content was 13% greater with translocation. Litterfall N flux was 42% lower with translocation whereas vegetation N increment was over 500% greater. Interestingly, translocation caused little change in the final forest floor N
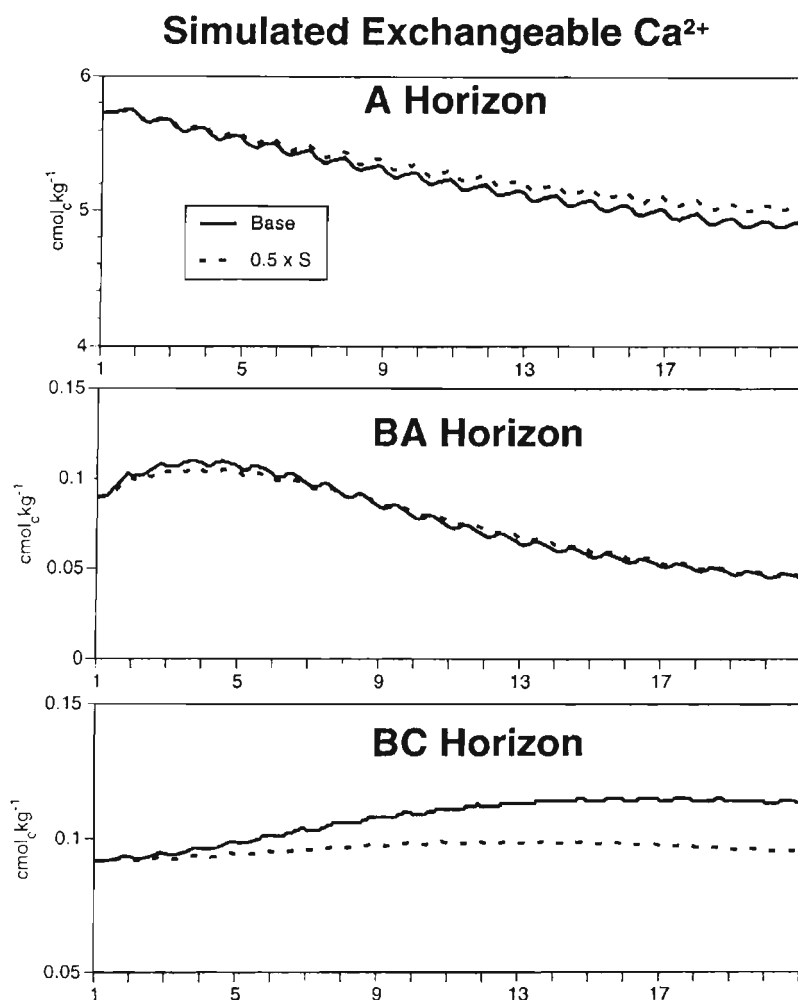
Simulated Exchangeable Ca²⁺

Fig. 7. Simulated exchangeable Ca^{2+} at Coweeta with no change and 50% reduction in S deposition (after Johnson et al., 1998). (Reprinted by permission of Springer-Verlag).

content, which was controlled more by decomposition rate than by litterfall N input. Perhaps most interesting of all was the response in N leaching, which was nearly 20 times greater with translocation. The greater N leaching with translocation was due to lower N immobilization in the litter and soil, and resulted in slightly lower net ecosystem N accumulation over the 24 years of simulation.

These results suggest that translocation greatly improves the efficiency with which trees can cycle N and is vital for the maintenance of tree N nutrition and growth. The importance of translocation has been well recognized by forest soil scientists and ecologists for many decades (see review by Nambiar and Fife, 1991) and does not represent a new insight gained from NuCM modeling. The suggestion that increased translocation may actually result in lower ecosystem-level N retention in the Walker Branch system is certainly counterintuitive. Whether this represents inherent flaws in model conceptualization and/or structure or an accurate prediction of real-world response cannot be known without field testing, which may be difficult. (How can translocation be reduced?)

Table 2
Simulated biomass and nitrogen contents after 24 years and simulated nitrogen fluxes over 24 years with and without translocation for a mixed deciduous forest at Walker Branch, TN (data adapted from Johnson, 1999)

Component	Translocation (kg ha⁻¹ of kg ha⁻¹ per year)	
	0%	47%
Biomass	200500	234800
N content		
Vegetation	569	645
Forest floor	255	248
Soil	2454	2365
Total ecosystem	3278	3258
N Fluxes		
Litterfall	805	468
Vegetation increment	21	110
Leaching	0.2	17

4. Suggestions for improvement of the NuCM model

NuCM, like all models, can be improved. We have identified the following potential changes based on the analysis above. However, we are well aware that adding complexity to an already complex model like NuCM may in fact lessen its value, and it may well be that these changes are not adviseable after a full analysis of the effects of such changes is made.

4.1. Simpler format of hydrologic output files to allow combination with other hydrological models

During the calibration of NuCMs hydrology to the Nordmoen site, it became clear that although NuCM produced a reasonable simulation of soil water percolation during the snow-free months, inaccurate simulation of percolation occurred during the winter months due to poor snow simulation and problems simulating a frost layer. Other hydrological models (e.g., the soil model (Jansson, 1991)) have proved better at simulating winter hydrology. By simplifying the hydrology output file format used in NuCMs biogeochemical simulation, the results from alternative hydrology models could be used when necessary.

4.2. Improvement of canopy sub-model

It would be desirable to simplify the present description of canopy processes (e.g., combining foliar exudation and leaching) and to include N transformations observed in Norway spruce (disappearance of inorganic N and production of organic N in the canopy).

4.3. Allowance of nutrient translocation for conifers

It is desirable and perhaps even necessary to include nutrient translocation in conifers. Work on this is underway at the Desert Research Institute at present.

4.4. Allowance for migration of roots and flexibility in root uptake among soil horizons

Currently, the user must specify what percentage of total uptake is to be taken from each horizon, by nutrient. This is a very sensitive calibration parameter: if the user causes too much of a given nutrient to be taken from a given horizon, deficiencies will develop and growth will slow down, even though there may be more than adequate supplies of the nutrient in question in other horizons.

4.5. Microbial nutrient uptake

Currently, there is no microbial nutrient uptake in NuCM. Over decadal time scales, this may suffice in that the net incorporation of nutrients in the microbial pool over long periods may be negligible. For simulations involving short-term responses (especially to N fertilization), however, it is important to include microbial uptake. The lack of microbial N uptake may also contribute to NuCMs relatively poor performance in intra-annual prediction of soil solution chemistry.

4.6. Inclusion of explicit simulated values for ecosystem carbon contents

NuCM does not explicitly simulate C accumulation, but uses C/N ratio in soils and simulates litter and vegetation mass. Only the vegetation mass is visible to the user as output.

5. Summary and conclusions

NuCM, like other models of its kind, has "too many knobs" to produce consistent predictions: there are usually far more parameters whose values are not known than there are values to apply to them. For example, NuCM requires that the user provide estimates of monthly uptake of nutrients (fraction of total annual uptake that occurs in each month). This information is almost never available, yet it is vital to the behavior of the model in an intra-annual time scale. Thus, users of NuCM must make educated guesses at these and other required but unknown input parameters. These educated guesses are nearly always incorporated into the initial calibration of the model, leaving open the possibility of getting a wide variety of model output values for a given site from different investigators and/or calibrations. In short, the results of NuCM modeling should not be considered unique for a given set of site conditions. This is not a flaw particular to NuCM; other similar models which operate on comparable time scales require this kind of information also and perhaps make fixed assumptions about it in the code that are invisible to the user.

NuCM has proven to be relatively "successful" in mimicking decadal scale changes in nutrient pools and soils (harvesting, species change, liming). It has not proven so successful in mimicking shorter-term (intra-annual) variations in soil solution chemistry, despite the fact that soil chemistry is one of the model's strong aspects. The reason for the latter is the lack of sufficient knowledge of controls on soil chemical and biological processes over shorter time periods. Like all models of its type, NuCM does not adequately portray several important processes simply because we do not have enough knowledge to write code for these processes. Prominent among these processes are N fixation and soil weathering. Nitrogen fixation is not accounted for in the model, and must be included in atmospheric deposition. The soil weathering module in NuCM is complex and probably as good as any available; however, soil weathering remains an elusive quantity to measure, let alone simulate. Soil weathering is not something that we can suggest as a need for improvement on, however, because our knowledge of weathering processes and, most importantly, how to quantify them are inadequate.

Despite its shortcomings, which we believe are no worse than any other model of its kind, we find that NuCM has considerable heuristic value. For example, the model has provided us with considerably insight regarding to the interactions of leaching and uptake in long-term nutrient budgets through the species change simulations. It has confirmed that our basic understanding of the reactions of lime with acidic soils can account for the long-term retention of Ca on the exchange sites in a numerically rigorous fashion. The mixture of "successes" and "failures" of the NuCM model, even within the same set of simulations, has in itself provided valuable insight into modeling: these "successes" and failures have shown that a model which fails to mimic short-term temporal variation in soil solution chemistry (as in the beech simulations) can nevertheless provide reasonable assessments of long-term changes in soils (species change and liming). In areas where no field data are available, NuCM has provided interesting hypothesis which can help guide research, such as its predictions regarding the rapidity and nature of soil change and the effects of translocation on ecosystem N retention and cycling.

Finally, the availability of the various "knobs" in the model that cause problems for consistent prediction at the same time offer great opportunities for the exploration of the interactions among nutrient cycling processes that may not exist in more "hard-wired" models. We have only begun to explore the implications of spatial (within litter and soil) and temporal (within season) variations in uptake that can be easily investigated with NuCM, for example. As we continue to test and explore the model, many more such opportunities will certainly arise.

Acknowledgements

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Statistical power in forest monitoring

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Abstract

Statistical power, the ability to detect a specified effect size (ES) (difference among treatments or change over time), *if* an effect exists, is low in most monitoring studies. *A priori* power analysis estimates the power of a particular experimental design and answers the following questions. (1) What is the smallest ES that can be detected? (2) How many samples are needed to detect effect sizes of biological or management significance? (3) What is the risk of wrongly accepting a false null hypothesis (Type II error) at a given risk of wrongly rejecting a true null hypothesis (Type I error)? When changes over time are gradual (as during silvicultural conversion of even-aged, second-growth forests to an uneven-aged condition) and resources are limited, power analysis helps allocate sampling effort to maximize the probability of detecting small effect sizes. The power of any particular experimental design can be enhanced by increasing sample size, increasing acceptable Type I error risk, and, in analysis of variance, specifying planned means comparisons. Alternatively, power can be increased by changing experimental design to reduce residual variance and/or increase ES. Use of parametric and one-tailed statistical tests also increases power. The forest monitoring program at Fort Lewis, a military installation, provides an example of the application of power analysis. Published by Elsevier Science B.V.

Keywords: Statistical power analysis; Monitoring; Error; Sample; Washington state

1. Introduction

Legal requirements for forest monitoring have increased dramatically in recent years. For example, in the Pacific Northwest, USA, the Northwest Forest Plan (Lujan et al., 1992) requires managers of Federal lands to monitor a host of plant, animal, and fungal species, and a variety of stand structure variables. Since money and personnel are limited, public land managers must be parsimonious when designing monitoring programs, yet have adequate experimental designs and reasonable confidence in their conclusions. Statistical power analysis can help by (1) quantifying the risks of making wrong conclusions and (2) indicating the minimum sampling effort needed to

detect relevant differences between treatments and changes over time in monitored variables.

Power analysis is particularly valuable when the expected differences are small. For example, in both Europe and the United States, there is growing interest in alternative silvicultural treatment of young, managed forests. Sometimes called "transformation", the objective is to take young, even-aged, usually single-species plantations and convert them to an uneven-aged, mutispecies condition. The general approach is to incrementally change stand characteristics through repeated entries, such as light thinnings. The trends in stand structure over time may be too gradual to detect on a 5- to 10-year time scale with the traditionally small sample sizes of most monitoring programs. This contrasts with the abrupt, large changes in stand structure that occur when old-growth or late-successional forests are partially cut, or any age of forest is clear-cut.

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This paper defines statistical power, describes the factors that influence it, and gives forest monitoring examples of the use of power analysis. These ideas are then applied to the monitoring program for Fort Lewis, a US Army installation in the Pacific Northwest.

2. Definition of statistical power

2.1. Null hypotheses and effect sizes

Whenever a statistical test is conducted, one or more alternative hypotheses are being evaluated against the null hypothesis, which is usually the nil hypothesis that effect size (ES) is zero. The definition of ES depends on the type of test. In the case of means comparisons, such as t-test and analysis of variance (ANOVA), ES is the difference between treatment means and the means of levels within treatments. In the context of forest monitoring, treatments are typically geographic location (e.g. watersheds, soil types, management units), human manipulation (e.g. silvicultural or site preparation methods) and/or time, while treatment levels are the individual locations, manipulations, and times.

In correlation and regression, respectively, ES is typically the difference between the actual value of the correlation coefficient (r) or the coefficient of determination (r^2) and a specified, null-hypothesis value, usually zero (Cohen, 1988; Borenstein et al., 1997). In forest monitoring, correlation or regression is often used to detect trends over time (i.e. across more than one time interval).

2.2. Statistical power and the risks of being wrong

When a statistical test is performed the decision being made is whether to accept or reject the null hypothesis. There are, however, actually four possible outcomes of a test, correctly or incorrectly accepting or rejecting the null hypothesis (Table 1). Incorrectly rejecting the null hypothesis is called Type I error. The maximum risk of committing Type I error that one is willing to accept, α, is traditionally set at 0.05. This means that if the statistical test were repeated many times, using different random samples from the same populations, about 5% of the outcomes would be significant if the null hypothesis was indeed correct.

Table 1
The four possible outcomes of a statistical test, and the associated risks of reaching wrong conclusions

"The Truth"	Statistical decision	
	Reject null hypothesis	Accept null hypothesis
Null hypothesis true	Type I error (α)	No error $(1-\alpha)$
Null hypothesis false	No error = power $(1-\beta)$	Type II error (β)

The actual Type I error risk, P, of a particular test must be equal to or less than α for the test to be considered "significant" and the null hypothesis rejected.

As numerous authors have pointed out in recent years (e.g. Toft and Shea, 1983; Peterman, 1990a,b) the other type of statistical error, Type II, or incorrectly accepting the null hypothesis, is seldom calculated. The maximum risk of Type II error that one is willing to accept is β. There is no traditional value for β, although some recommend 0.20 (e.g. Cohen, 1988). Yet it is still common in the literature for authors to uncritically accept the null hypothesis when a statistical test is nonsignificant. As Freiman et al. (1978), Hayes (1987), Peterman (1990b), and others have shown, Type II error is often quite large, especially when replication (sample size or n) is small.

Power is defined in Table 1 as $1 - \beta$. It is the probability that a statistical test will correctly reject the null hypothesis. Stated another way, it is the likelihood that a statistical test will detect a particular ES, assuming it exists. The value of β, for all statistical tests, is affected by n, ES, α, and the standard deviation (s) of the data. These relationships are demonstrated by power curves.

Fig. 1 is an example of power curves for a hypothetical comparison of shrub cover under conifer forest canopies. There are two treatment levels, control (no thinning) and moderate thinning (25% stem removal), as might occur during transformation from even-aged to uneven-aged condition. Cover is measured on replicate, 100 m² plots in each treatment (n = total number of plots per treatment). Mean control cover is $16 \pm 5\%$ (mean $\pm s$). ES is the relative difference (positive or negative) in mean percent cover between the control and thinned treatments. Power is calculated for an unpaired t-test, as a function of n, α, ES, and s, using SamplePower software (Borenstein et al., 1997). Initially, power increases rapidly with increased n, but
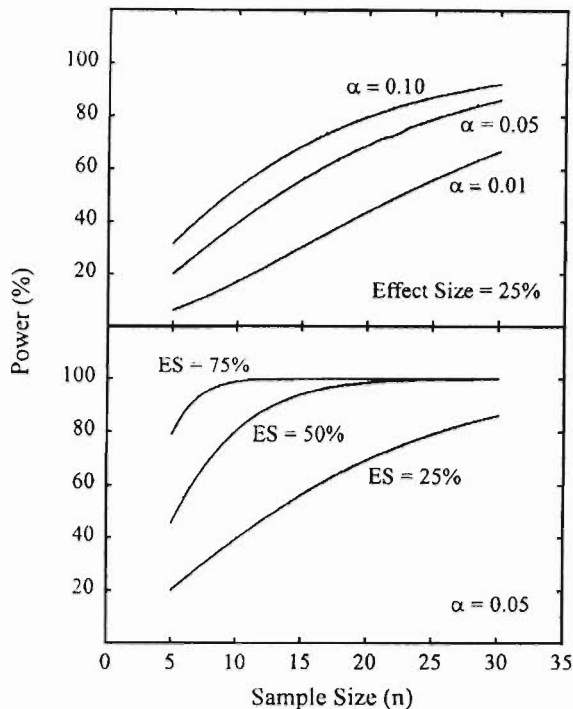
Fig. 1. Power curves showing the effects of sample size (n), ES, and maximum acceptable risk of Type I error (α) on statistical power (1 − β) for a comparison of shrub cover in thinned and unthinned conifer forests. Calculations are for unpaired t-tests. Effect size is relative, i.e. ES = 25% represents an increase or decrease in shrub cover of 4% from the control value of 16% (4/16 = 0.25).

diminishing returns apply: above a threshold n there is little further increase in power (Fig. 1). As α increases, power increases (Fig. 1, top panel), i.e. increasing the risk of Type I error decreases the risk of Type II error. As ES increases, so does power (Fig. 1, bottom panel), i.e. large differences between means are easier to detect than small differences.

The formulae for calculating power depend on the type of statistical test. The general approach is to standardize ES in dimensionless terms (Cohen, 1988). For the paired t-test, this means dividing ES (mean difference between paired observations) by the standard deviation of the differences (s_{diff}). For ANOVA, this means dividing the between-groups standard deviation ($s_{between}$) by the within-groups standard deviation (s_{within}). For correlation, no standardization of r is necessary. For regression, ES = explained variance ($s^2_{regression}$) divided by error

variance (s^2_{error}). Using standardized ES, the value of the test statistic (t, F, etc.) is calculated that would yield a significant result for specific values of n, ES, and α. Finally, the proportion of studies (if the study was to be repeated many times) that would yield a significant test statistic, i.e. power, is determined.

2.3. Methods to increase power

In most ecological studies, including forest monitoring programs, replication is low (n = 10 or less), so even modest increases in n may yield large increases in power (Fig. 1). Therefore, when resources are limited, it is usually better to measure fewer variables with more replication and higher power than to measure more variables with less replication and lower power.

Increasing α is an option not often considered. The seemingly magical formula, "α = 0.05", is merely a long-established convention; most people agree that 1-in-20 odds of being wrong are favorable odds. But this fails to consider either the magnitude of β or the relative costs of Type I versus Type II error (Toft and Shea, 1983; Peterman, 1990a,b). Ideally, one would like both α and β to be small (e.g. ≤0.05) but this may only be achievable with extraordinarily large sample sizes. The alternative is to increase acceptable Type I error risk (e.g. α = 0.10) until the ratio of Type I to II error is similar to the ratio of the costs of Type I and Type II errors (Peterman, 1990a).

Consider the following example, a public land manager wants to know if the current practice of removing snags during logging operations is reducing the population size of a threatened woodpecker species. A study is established to compare population sizes of woodpeckers in 20 replicate stands, 10 with snags removed and 10 with snags left in place. Mean woodpecker population size is greater in stands with snags left in place than in stands with snags removed. The means are compared with an unpaired t-test. If the test is significant, the manager concludes that leaving snags increases woodpecker population sizes. The manager decides to leave snags from now on, foregoing the monetary value of the wood in sound snags and increasing fire danger due to more fuel accumulation. If the manager's conclusion is wrong, these are the (relatively minor) costs of a Type I error. If the test is non-significant, the manager concludes that leaving

snags has no effect, and continues the policy of removing snags. But if 5 years later, monitoring indicates that woodpecker populations are decreasing in stands where snags are still removed, a Type II error has occurred, there really is an effect of logging on woodpeckers. A regulatory agency halts all logging operations until populations increase back to a certain level, a very high cost indeed. In this example, the manager probably should have designed the monitoring program so that β was less than α.

Planned means comparisons can be used in analysis of variance (ANOVA) to increase power. One specifies an alternative hypothesis that is more specific than the general alternative hypothesis associated with ANOVA, i.e. that *some* kind of ES exists. For example, the means of three treatment levels can differ in nine ways ($\overline{X}_a > \overline{X}_b > \overline{X}_c$, $\overline{X}_a = \overline{X}_b > \overline{X}_c$, $\overline{X}_c > \overline{X}_a > \overline{X}_b$, etc., where $\overline{X}$ = mean) but a significant ANOVA does not tell you which of these alternatives is correct. The value of α for the overall ANOVA (e.g. 0.05) applies to each planned comparison. To justifiably using planned means comparisons, one must have good reasons for expecting particular outcomes. Otherwise, one is restricted to doing unplanned means comparisons after the results of the ANOVA are known. In this case, α for the overall ANOVA applies to all comparisons combined (so-called "experimentwise" α), so α, and thus power, is less for each individual comparison than for the overall ANOVA (Sokal and Rohlf, 1981).

While the actual value of ES is a property of the population(s) being studied, one can lower *standardized* ES by altering the experimental design to reduce residual variance, i.e. the proportion of variance attributable to random error and uncontrolled factors (e.g. s_{within} in ANOVA and s^2_{error} in regression). For example, subsamples may be taken within each unit of replication and averaged to yield a single value for each replicate. Alternatively, the criteria for selection of units of replication can be tightened so that they are more uniform in their characteristics. Other techniques to reduce residual variance include use of randomized blocks in ANOVA, "pairing" of data (as in paired *t*-tests and repeated-measures ANOVA), and analysis of covariance.

It is also possible to change the experimental design so that actual ES is increased. One way to accomplish this is to increase the strength of the non-control treatments. For example, in an experiment comparing bole volume growth in stands subjected to varying intensities of thinning, one could increase the differences among treatments in the proportion of basal area removed from the initial values of 0, 15, and 30% to 0, 25, and 50%. Actual ES can also be increased by measuring a response variable that is more sensitive to the treatment than the variable initially selected.

The use of parametric statistical tests is preferred over non-parametric tests because they have higher power for any given experimental design. If the data do not meet the assumptions of parametric tests — normally distributed with equal variances among groups — they can often be transformed to meet the assumptions (Sokal and Rohlf, 1981).

For statistical tests which are directional, i.e. the effect may be either an increase or a decrease (but not both) from the null hypothesis, a one-tailed test has more power than a two-tailed test (the effect can be in either direction). One-tailed tests are justified in two instances. First, change in one direction is physically impossible. For example, no natural regeneration is occurring in a stand subjected to an individual-tree selection harvest to initiate conversion. Will mowing the dense brush encourage regeneration? Clearly, negative regeneration cannot occur, any possible effect of mowing must be in one direction only, an increase above zero (no regeneration). Second, change in one direction is functionally equivalent to no change (Borenstein et al., 1997). For example, a forest products company wants to know if adoption of more costly group selection will decrease watershed-level soil erosion compared to the current system of two-step shelterwood. If group selection either increases or has no effect on soil erosion, the consequence for the company is the same, stick with less-expensive shelterwood.

A hypothetical example demonstrates how to choose an appropriate method to increase power. Suppose that the question of interest is the effect on soil compaction of different methods of yarding logs to landings when steep slopes are logged. An experiment is designed as follows, 15 stands of similar slope, species composition, commercial wood volume, and soil type are selected for study. Five stands are randomly allocated to each of three treatment levels, rubber-tired skidders, tracked skidders, and high-lead cable yarding. In each stand, about 25% of the wood volume is removed. Following logging, soil density is

Table 2
Power analysis for one-way ANOVA on soil compaction, comparing stands logged using different methods of yarding logs to landings[a]

Mean penetration depth (cm) ($\pm s$)						
Method			Power (%)			
Rubber-tired skidder	Tracked skidder	Cable yarding	Base case	$n = 10$	$\alpha = 0.10$	ES = 3.0 cm
15.0 (5.3)	12.0 (2.2)	9.0 (2.6)	51	88	66	86

[a] The effects of doubling sample size (n), doubling the maximum acceptable risk of Type I error (α), and increasing effect size (ES) by 50% are shown. For the base case, $n = 5$, $\alpha = 0.05$, and ES = 3 cm. For simplicity, ES is symmetrical about the mean of all groups.

measured with a penetrometer at 10 randomly chosen points within each stand. Penetration depths are averaged within each stand, since stands are the unit of replication. Means are compared with one-way, fixed-effects ANOVA. The test, given $\alpha = 0.05$, is nonsignificant ($P = 0.07$) and the null hypothesis of no difference among yarding methods in degree of soil compaction cannot be rejected.

Table 2 shows the data and the results of power analysis using SamplePower. Power is only 51% for the base case ($n = 5$, $\alpha = 0.05$, absolute ES = 3.0), so the risk of Type II error is 49%, very high indeed. Although doubling n yields a large increase in power, it is expensive and time-consuming to log more stands, so doubling α is the next-best option. Since, ES is an inherent property of this particular experiment, it cannot be changed after the fact. However, the study could initially have been done with greater volume removal to accentuate any differences among yarding methods that might exist. Yet another, low-cost option, would be to increase the subsampling within stands from 10 to 50 penetrometer measurements. This would reduce s_{within} in the ANOVA, thus increasing ES and power.

Designing studies with adequate power requires estimates of s and ES. But what if the study is being planned and no data have been collected yet? In order of preference, I recommend the following data sources (1) a pilot study with lower n and/or a shorter time-frame than the full study (Green, 1989); (2) a previous study (done by the investigator or published by others) that measured the same variables and used similar methods and experimental design to the planned study; and (3) simulation modeling (e.g. Brady et al., 1995). An initial, ballpark estimate of the variability of the data being studied is better than no estimate.

2.4. Minimum sample sizes and detectable effect sizes

Another use of power analysis is to calculate minimum n to detect specified differences between means or specified changes over time. If n is too small, important effects will not be detected. If n is too large, time and resources are wasted.

For any given n, α, β, and s, there is a minimum detectable effect size (MDES), the smallest ES that can be detected by a particular statistical test (Anderson et al., 1996). If α, β, s, and MDES are known, minimum n can be calculated. Bernstein and Zalinski (1983) recommend determining what constitutes a "relevant" ES, then designing a sampling program to detect it with a specified power and α. This is done by making the relevant ES equivalent to MDES and solving for n.

Relevant ES is the minimum ES (absolute or relative) that is of ecological or management significance. As an example of ecological significance, a 10% increase in ground layer cover may be of little consequence to animals that feed or nest in that layer, whereas a 50% increase would substantially increase their populations. As an example of management significance, if the 5-year wood volume increment of forest stands receiving nitrogen fertilizer is only 2% higher than control stands, this would hardly justify the cost of fertilization, but it might be cost-effective if the increase is 10%.

2.5. Uses of power analysis

Much of the literature on power deals with retrospective analyses of already published data to determine whether or not the author(s) justifiably accepted the null hypothesis. Peterman (1990a) refers to this as

a posteriori power analysis. However, the most valuable application of power analysis is *a priori*, i.e. estimating the power and MDES of studies that have not yet been carried out (Peterman, 1990a).

3. The Fort Lewis example

I will now apply the principles discussed above to an actual example of forest monitoring. I will demonstrate how a priori power analysis can provide estimates of power, MDES, and minimum *n* for a particular experimental design, and for a variety of different stand structure variables.

3.1. Environmental setting

Fort Lewis is a 35,000 ha US Army installation, located between Tacoma and Olympia in Washington State (latitude 122°35″W, longitude 47°05″N). The majority of the installation is underlain by glacial outwash, and most of the rest by glacial till, moraine, and undifferentiated drift (Noble and Wallace, 1966; Walters and Kimmel, 1968). As a result, excessively drained, gravelly soils occupy nearly 90% of the land area (Anderson and Ness, 1955; Pringle, 1990).

Approximately 20,600 ha of the installation are occupied by conifer forests, 800 ha by hardwood forests, 1095 ha by Oregon white oak (*Quercus garryana* Dougl. ex Hook.) woodlands, 8250 ha by grasslands (locally called "prairies") and 1450 ha by non-forested wetlands.

The occurrence of forest types on the landscape is a function of soil type, topography, and past land-use history (Ahrens, 1998). The conifer forests consist of three major forest types: (a) dry forests on prairie soils (DP forests; 4100 ha or 19% of the total forest area), dry forests on forest soils (DF forests; 4900 ha or 23% of area), and moist forests on forest soils (MF forests; 4500 ha or 21% of area).

DP forests consist of "colonization" Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) stands occupying former prairies, and underlain by very rocky, excessively drained soils formed on glacial outwash plains. These "first-generation" stands lack "legacy", i.e. snags, logs, and large, live trees left over from previous stands. The soils, mostly in the Spanaway series, have a thick, organic-rich A horizon that

formed under prairie vegetation (Pringle, 1990). Comparison of 1853 and 1870 land surveyor notes with modern vegetation suggests that at least 6500 acres of former prairie are now DP forest, the result of exclusion of Native American-set fires (Public Forestry Foundation, 1996).

DF forests consist of Douglas-fir stands growing in areas that historically were forest, and are underlain by rocky, excessively or well-drained forest soils on hilly areas of glacial till and moraine. Historically, these forests had similar species composition, but were much more open (lower tree density) than today, probably due to underburns when prairie fires entered the forests (Public Forestry Foundation, 1996). Soils are mostly in the Everett series (Pringle, 1990).

MF forests consist of conifer stands growing on soils moist enough for western redcedar (*Thuja plicata* Donn ex. D. Don), western hemlock (*Tsuga heterophylla* [Raf.] Sarg.), and other shade-tolerant species to occupy the overstory, and occur on moderately drained forest soils on hilly areas of glacial till and moraine. Soils are mostly in the Alderwood, Tenino, and Sinclair series (Pringle, 1990). Successional stands are often dominated by red alder (*Alnus rubra* Bong.).

3.2. Management setting

Most of the forests were cleared in the 20th century. The portion south of the Nisqually River was harvested in the 1940s, and large areas were clearcut in the 1960s and 1970s. Thus, modern-day forests are mostly even-aged, 45–100 years old, and dominated by Douglas-fir.

Commercial forests at Fort Lewis occupy 17,563 ha or 78% of the total forest area. Each stand on Fort Lewis is classified into a management unit, representing a combination of stand age, basal area, forest type, and past management history. Most stands (12,844 ha or 73% of the commercial forests) are commercial conifers ≥45 years old, with low, intermediate, or high basal areas (<28, 28–41 and >41 $m^2 ha^{-1}$, respectively). Past management varies, but the majority (8529 ha or 66%) of these commercial conifer stands have had one or two light thinnings.

Forest management at Fort Lewis is guided by a Forest Management Strategy (Public Forestry Foundation, 1996). Its primary objective is to develop and maintain late-successional forests. Goals include

uneven-aged, multispecies stands, with multiple, patchy canopy layers and increased numbers and sizes of snags and logs. The complementary Northern Spotted Owl Habitat Management Plan for Fort Lewis (Bottorff, 1994) quantifies many of these goals. In 1992, most of installation's forests were designated critical habitat for the owl (*Strix occidentalis caurina* [Merriam]) by the US Fish and Wildlife Service, so the Army must manage for habitat features important to feeding, roosting, and nesting by owls.

Although natural succession would eventually create late-successional forests, Fort Lewis is accelerating the development of late-successional stand structures with silvicultural manipulation. Our primary tool is variable-density thinning (VDT). In traditional thinning, which was practiced at Fort Lewis until the early 1990s, the objective is for the residual stand to have more evenly spaced trees of more similar size than the pre-thinning stand. In VDT, the objective is for the residual stand to have more irregular tree spacing with more variation in tree size than the pre-thinning stand. This is accomplished by varying the density of tree removal across the stand and thinning across all commercial size classes.

3.3. Monitoring program

The major questions to be addressed are: (1) What is the influence of VDT on forest stand 14 structure? (2) How does forest stand structure differ among forest types?

Stands are chosen from the 429 commercial conifer stands that are ≥45 years old and have received one or two traditional thinnings in the past. Each stand is classified into one of three forest types (DP, DF, MF), two age classes (45–65 years old and >65 years old), three basal areas, and one or two previous thinnings, a total of 36 combinations. This is clearly an unrealistic number of combinations to monitor with adequate replication, so the following assumption was made: over time, VDT will cause stands of differing age, basal area, and number of previous thinnings to converge in stand structure. In other words, in the long run, differences between stands will most likely be attributable to differences in forest type.

The experimental design is two-way, repeated-measures ANOVA, where the treatments are forest type and time, and levels within treatments are MF, DF, DP

(forest type) and successive 5-year intervals (time). The unit of replication is individual forest stands, which are repeatedly sampled over time. Sample size is 12 stands per forest type (36 stands total). Sub-replication consists of sample plots laid out along 70.4 m × 70.4 m grids within each stand. Snags and logs are measured in every plot, all other variables in alternating plots. The doubled subreplication for snags and logs, which are very patchily distributed in space, reduces their s_{between} in the ANOVA. Commercial trees are censused with variable-radius plots, all other variables in nested, fixed-radius circular plots.

The frequency of sampling will be every 5 years. However, rather than measuring all 36 stands every 5 years, the sampling scheme is tied to the timber sale schedule. Since the effects of VDT are of interest, and the interval between thinnings for each stand is ≈10 years, the 5-year frequency for each stand consists of just before its first VDT, 5 years after its first VDT, 10 years after its first VDT (just before its second VDT) and so forth, for each successive round of VDT. Thus, each year, the list of commercial conifer stands scheduled to be sold for VDT is examined and 2–3 stands are randomly chosen for sampling within each forest type (6–7 stands year^{-1}). After the first 5 years of the inventory, all 36 stands will have been measured, and repeat measurements will begin.

Based on existing knowledge of forest stands and expected future trends, I have developed a priori hypotheses as to how the different forest types should rank for each variable (Table 3). These planned comparisons will permit use of the same α for individual comparisons of forest types and inventory years as for the overall ANOVA, thus increasing the power of the comparisons. The threshold α for significance will be 0.10.

3.4. Power, MDES, and minimum sample size estimates

What are the smallest differences between forest types and the smallest changes over time that this new stand structure inventory can detect? Are these sufficient to assess progress towards the goals of the Forest Management Strategy and Owl Plan? Power analysis provides guidance on this issue.

Since the inventory is brand-new, there are no pilot study data for use in power analysis. Instead, I use data

Table 3
A priori hypotheses for differences between forest types and changes over time in forest stand structure at Fort Lewis[a]

Stand attribute	Hypotheses		Relevant ES (%)
	Differences among forest types	Changes over time	
Canopy cover	MF > DF > DP	Decrease	20
Basal area	MF = DF = DP	No change	20
Tree density[b]	MF = DF > DP	Decrease	20
Sapling density[c]	DP > DF = MF	Increase	50
Average DBH[d]	DP > MF > DF	Increase	20
Total log length[e]	MF > DF > DP	Increase	50
Snag density[f]	MF > DF > DP	Increase	100

[a] MF: moist forests on forest soils; DF: dry forests on forest soils; DP: dry forests on prairie soils. Relevant ES is the minimal effect size considered to be of biological or management significance.

[b] Stems >10 cm DBH.

[c] Stems >45 cm tall but ≤137 cm tall.

[d] Diameter at breast height (1.47 m above ground).

[e] Cumulative length of all logs ≥50 cm midpoint diameter and ≥6 m long.

[f] Snags ≥2 cm DBH and ≥3 m tall.

from a previous stand inventory conducted between 1996 and 1998. This inventory measured many of the same variables, with similar plot dimensions, as the new inventory. Plot spacing within stands was 141 m × 141 m for all variables, including logs and snags. About 41 of these stands are unambiguously classified as DP, 16 as DF, and 18 as MF, a total of 75 stands.

ANOVAs on a representative set of variables (also to be measured in the new inventory) indicate that forest types differ significantly ($\alpha = 0.10$, P ranges from 0.07 to <0.001) for all variables. However, the specific patterns by which they differ is not the same for each variable (a posteriori means comparisons at $\alpha = 0.05$ for all comparisons combined). For example, canopy cover is significantly greater in MF than in DF stands, and in DF than in DP stands. On the other hand, average tree diameter at breast height (DBH) is significantly less in DF stands than in either DP or MF stands, and does not differ between the latter two stand types.

This raises problems for power analysis. Recall (Section 2.2) that standardized ES is $s_{between}/s_{within}$. Assume that the average relative difference between groups is 20%. How this difference is divided up has a strong effect on ES. If the difference is evenly split (i.e. MF is 10% greater than DF, which is 10% greater than DP), $s_{between}$ is maximized, and so are ES and power. If the difference is between only two of the

groups (e.g. MF is 20% greater than DF, which is the same as DP), $s_{between}$ is minimized, and so are ES and power. Therefore, a power analysis for ANOVA is useful only if I assume that differences between forest types maintain the same relative proportions to each other through time.

An alternative approach is to look at each possible pairwise comparison by doing power analysis for a set of unpaired t-tests. In each case, the question is what the relative MDES (% difference) is for power = 90%, $\alpha = 0.10$, and $n = 12$ (the sample size for the new inventory), and assuming the same standard deviation for each forest type in the new inventory as were found for the previous inventory.

Carrying out the analysis with SamplePower software (Borenstein et al., 1997) yields the results shown in Table 4. Out of 21 possible comparisons, only $3/21 = 14\%$ are expected to have power ≥90%, and $5/21 = 24\%$ are expected to have power <20%. Relative MDES ranges from 21 to 100%. I expected power to be lower and relative MDES higher for logs and snags, which are very patchily distributed (many plots have none, some have a lot), than for other variables, but this was not generally the case (Table 4).

The largest differences between forest types found by the previous inventory are 24% (canopy cover), 20% (tree basal area), 30% (tree density), 83% (sapling density), 28% (average DBH), 71% (total log length), and 96% (snag density). The minimum n to

Table 4
Power analysis of comparison of stand structure attributes between forest types, using data from a previous-inventory[a]

Variable	Mean (S.D.)			Power of comparison (%)			Relative MDES (%)			Minimum sample size
	MF	DF	DP	MF vs. DF	DF vs. DP	MF vs. DP	MF vs. DF	DF vs. DP	MF vs. DP	
Canopy cover (%)	70 (16)	61 (14)	53 (16)	42	34	80	27	31	28	17
Tree basal area ($m^2 ha^{-1}$)	36.2 (13.9)	32.6 (11.2)	29.0 (9.3)	18	22	43	43	38	40	46
Tree density (stems ha^{-1})	234 (113)	268 (135)	188 (110)	17	46	26	65	56	58	41
Sapling density (stems ha^{-1})	337 (347)	170 (176)	459 (471)	42	61	18	100	72	60	27
Average DBH (cm)	51 (10)	41 (11)	57 (11)	74	95	35	25	25	21	10
Total log lenght ($m^2 ha^{-1}$)	74 (85)	69 (43)	20 (30)	11	94	94	54	72	70	10
Snag density (stems ha^{-1})	0.52 (1.19)	0.05 (0.22)	0.37 (1.11)	37	25	12	96	95	79	12

[a] Variables and forest types are defined in Table 3. Power is computed for unpaired t-tests with $\alpha = 0.10$ and $n = 12$. MDES: minimum detectable effect size for $\alpha = 0.10$ and power $= 90\%$. Minimum sample size is that required to detect the largest difference between means of forest types in the previous inventory at $\alpha = 0.10$ and power $= 90\%$.

detect these differences is similar to the new-inventory n (=12) for average DBH, total log length, and snag density, and not much greater than the new-inventory n for canopy cover (Table 4). Detecting these differences for the other variables would, however, require much larger n.

A similar approach is taken for power analysis of changes over time. What is relative MDES for each stand structure variable if it is measured in one year, then again 5 years later? Since the same stands are measured over time, paired t-tests are used, with $\alpha = 0.10$ and $n = 36$ (all stands combined). For each variable, standard deviation is assumed to have the same value as for the previous-inventory ($n = 75$ stands) and to be equal at both measurement times. It is also assumed, for purposes of calculating the standard deviation of the difference between sampling times, that the correlation between measurements at the two sampling times is 0.80 (Borenstein et al., 1997, p. 61).

Results of SamplePower analysis are shown in Table 5. Because of the larger n (36 versus 12), relative MDES values are smaller than in the previous analysis for all variables except snag density, and they are sufficient to measure those ESs considered to be biologically or managerially relevant (Table 3) for all variables except snag density. Note, however, that snag density relative MDES (181%) is only 12% of the relative change (1460%) from the current condition (0.32 stems ha^{-1}) needed to meet the Owl Plan goal of 5 stems ha^{-1}, so the new inventory is quite capable of

Table 5
Power analysis for comparison of stand structure attributes over time, using data from a previous inventory[a]

Variable	Mean (S.D.) of current value	MDES (%)
Canopy cover (%)	59 (17)	8
Basal area (m² ha⁻¹)	31.5 (11.2)	11
Tree density (stems ha⁻¹)	216 (120)	17
Sapling density (stems ha⁻¹)	368 (409)	35
Average DBH (cm)	52 (12)	8
Total log length (m ha⁻¹)	43 (56)	23
Snag density (stems ha⁻¹)	0.32 (1.01)	181

[a] Variables are defined in Table 3. Current values represent all stands combined ($n = 75$). MDES: minimum detectable effect size, calculated for paired t-tests with power = 90%, $\alpha = 0.10$, and $n = 36$.

Table 6
Power analysis for comparison of stand structure attributes of individual forest types over time, using data from a previous inventory[a]

Variable	Relative MDES (%)		
	MF	DF	DP
Canopy cover (%)	13	13	17
Basal area (m² ha⁻¹)	22	19	18
Tree density (stems ha⁻¹)	27	28	33
Sapling density (stems ha⁻¹)	58	59	59
Average DBH (cm)	15	15	15
Total log length (m ha⁻¹)	65	35	85
Snag density (stems ha⁻¹)	129	260	70

[a] Variables and forest types are defined in Table 3. Current values for each forest type are shown in Table 4. MDES: minimum detectable effect size, calculated for paired t-tests with power = 90%, $\alpha = 0.10$, and $n = 12$.

detecting incremental change towards this goal. The relative MDES for total log length is also well below the relative changes necessary to attain Owl Plan goals — total log length 90 m ha⁻¹ in DF and DP stands, 180 m ha⁻¹ in MF stands — which are 109 and 319% greater, respectively, than the current values shown in Table 5.

Repeating the analysis of Table 5, but for detecting change in individual forest types over time, yields higher values of relative MDES compared to the previous analysis, for all variables except snag density (Table 6). This is not surprising, since $n = 12$ instead of 36, as in the previous analysis. The relative MDESs suggest that the new inventory should consistently detect relevant ESs (Table 3) for canopy cover, basal area, and average DBH, and, in some forest types, relevant ESs for total log length and snag density.

4. Discussion

4.1. Assumptions of the analyses

The power analyses in this paper make the usual assumptions of parametric statistics: (1) random sampling, (2) normally distributed data, and (3) equal variances among groups. The data from the previous inventory meet the first assumption (stands are randomly selected), but sapling density and snag density

(Table 5) clearly fail the second assumption. Some variables in the new inventory will also likely not meet all assumptions. However, at least in the case of *t*-tests, moderate departures from normality have minor effects on power (references in Anderson et al., 1996, p. 18).

4.2. Implications for Fort Lewis forests

Based on a priori power analyses, the Fort Lewis monitoring program will not always have high power (≥90%) to detect differences in stand structure between forest types (Table 4). Changes in stand structure over time, for the three conifer forest types combined, will be detected with high power for all variables except snag density (Table 5), but not all relevant changes over time for individual forest types will be detectable with high power (Table 6). It should be noted that the power analyses in Tables 5 and 6 compare only one sampling interval. Trends in stand structure over two or more sampling intervals are detectable with higher power than a single interval of change (Gerrodette, 1987; Peterman, 1990a; Taylor and Gerrodette, 1993).

4.3. Recommendations for forest monitoring

A priori power analysis should be performed before new monitoring programs begin. The first step should be careful consideration of expected differences between treatments and treatment levels. This focus on specific hypotheses will increase the power of means comparisons in ANOVA and, when justified, *t*-tests, correlations, and other statistical tests which can have directional outcomes. The minimum relevant ES for each comparison should also be designated in advance, based on an understanding of the biology and current management of the forest of interest and its component species. These can then be compared with MDESs from power analysis to see if the proposed monitoring can detect relevant ESs.

If possible, a pilot study of limited duration and/or reduced sampling should be conducted to generate initial estimates of population means and variances for use in power analysis. Alternatively, data from previous studies in similar forests and with similar methods should be used. In the complete absence of data, simulation modeling may be used.

Changes in forest stand structure during conversion of even-aged forests to uneven-aged conditions will generally be gradual, accomplished by multiple stand entries over many years. Power analysis is particularly crucial for monitoring these changes because, if sample sizes are too low, small but meaningful ES may not be detected.

When ES is small, the best way to increase low power is usually to increase sample size. However, it may sometimes be less expensive to alter the experimental design to minimize residual variance and/or increase actual ES, as described in Section 2.3. When either of these two approaches is logistically impossible, one should consider increasing α above the conventional value of 0.05. In doing so, consider the relative costs of Type I versus Type II errors.

Alternatively, consider increasing the minimum relevant ES; perhaps you do not need to detect such small differences between treatments or changes over time as originally planned.

Acknowledgements

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Soil carbon cycling in a temperate forest: radiocarbon-based estimates of residence times, sequestration rates and partitioning of fluxes

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Abstract. Temperate forests of North America are thought to be significant sinks of atmospheric CO_2. We developed a below-ground carbon (C) budget for well-drained soils in Harvard Forest Massachusetts, an ecosystem that is storing C. Measurements of carbon and radiocarbon (^{14}C) inventory were used to determine the turnover time and maximum rate of CO_2 production from heterotrophic respiration of three fractions of soil organic matter (SOM): recognizable litter fragments (L), humified low density material (H), and high density or mineral-associated organic matter (M). Turnover times in all fractions increased with soil depth and were 2–5 years for recognizable leaf litter, 5–10 years for root litter, 40–100+ years for low density humified material and >100 years for carbon associated with minerals. These turnover times represent the time carbon resides in the plant + soil system, and may underestimate actual decomposition rates if carbon resides for several years in living root, plant or woody material.

Soil respiration was partitioned into two components using ^{14}C: recent photosynthate which is metabolized by roots and microorganisms within a year of initial fixation (Recent-C), and C that is respired during microbial decomposition of SOM that resides in the soil for several years or longer (Reservoir-C). For the whole soil, we calculate that decomposition of Reservoir-C contributes approximately 41% of the total annual soil respiration. Of this 41%, recognizable leaf or root detritus accounts for 80% of the flux, and 20% is from the more humified fractions that dominate the soil carbon stocks. Measurements of CO_2 and $^{14}CO_2$ in the soil atmosphere and in total soil respiration were combined with surface CO_2 fluxes and a soil gas diffusion model to determine the flux and isotopic signature of C produced as a function of soil depth. 63% of soil respiration takes place in the top 15 cm of the soil (O + A + Ap horizons). The average residence time of Reservoir-C in the plant + soil system is 8±1 years and the average age of carbon in total soil respiration (Recent-C + Reservoir-C) is 4±1 years.

The O and A horizons have accumulated 4.4 kgC m^{-2} above the plow layer since abandonment by settlers in the late-1800's. C pools contributing the most to soil respiration have short enough turnover times that they are likely in steady state. However, most C is stored as humified organic matter within both the O and A horizons and has turnover times from 40 to

34

100+ years respectively. These reservoirs continue to accumulate carbon at a combined rate of 10–30 gC m^{-2} yr^{-1}. This rate of accumulation is only 5–15% of the total ecosystem C sink measured in this stand using eddy covariance methods.

Introduction

Well drained temperate forest soils in the northeastern United States have accumulated carbon (C) over the past century as forest has regrown over former fields and pastures. The rate at which mid-latitude forest vegetation and forest soils are still accumulating C and can act to ameliorate future anthropogenic inputs of CO_2 to the atmosphere is still uncertain. The capacity for ecosystems to store CO_2 depends both on their productivity and the residence time of C (Thompson et al. 1996). Hence, the average time between fixation of C by photosynthesis and its return to the atmosphere by respiration or decomposition is an important parameter for determining the timing and magnitude of C storage or release in response to disturbances like climate or land use change (Fung et al. 1997).

Eddy flux tower measurements made since 1990 in a temperate deciduous forest in central Massachusetts (Harvard Forest) show consistent net ecosystem uptake of C averaging nearly 200 gC m^{-2} yr^{-1} (Wofsy et al. 1993; Goulden et al. 1996). Interannual variability in the rate of net C storage has been linked to climate (Goulden et al. 1996). The Harvard Forest is growing on land used for agriculture or pasture in the 19th century and was damaged by a hurricane in 1938. Net carbon storage in a forest recovering from these disturbances is not surprising. However, the partitioning of C storage among vegetation and soils at this site is unknown, as is the potential for C storage rates to change in the future as recovery from disturbance progresses.

The goal of this work is to quantify the below ground carbon cycle in well drained soils that dominate the footprint of the eddy flux tower at the Harvard Forest. We use radiocarbon (^{14}C) measurements in soil organic matter (SOM) and CO_2 to quantify the residence time of C in the plant + soil system and to determine the contribution of well-drained soils to the net sink measured by Wofsy et al. (1993) and Goulden et al. (1996). We also partition total soil respiration into two components using ^{14}C: (1) root respiration and microbial metabolism of recent photosynthate within a year of initial fixation (Recent-C), and (2) CO_2 derived from microbial decomposition of SOM that resides in the soil longer than a year (Reservoir-C).

Radiocarbon measurements of SOM and CO_2 are an extremely useful tool to determine the dynamics of soil carbon. ^{14}C produced by atmospheric weapons testing in the early 1960's (i.e. 'bomb C') is used as an isotopic
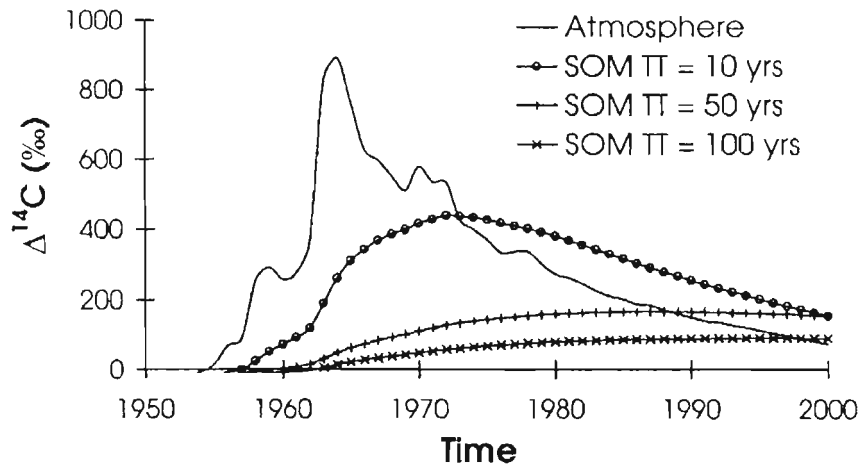
Figure 1. The time record of ^{14}C in the atmosphere (Northern Hemisphere) based on grapes grown in Russia (Burchuladze et al. 1989) for 1950–1977 and direct atmospheric measurements for 1977–1996 (Levin & Kromer 1997). We express radiocarbon data here as $\Delta^{14}C$, the difference in parts per thousand (per mil or ‰) between the $^{14}C/^{12}C$ ratio in the sample compared to that of a universal standard (oxalic acid 1, decay-corrected to 1950). All samples are corrected for mass-dependent isotopic fractionation to -25‰ in $\delta^{13}C$. Expressed in this way, $\Delta^{14}C$ values greater than zero contain bomb-produced radiocarbon, and those with $\Delta^{14}C$ less than zero indicate that carbon in the reservoir has, on average, been isolated from exchange with atmospheric $^{14}CO_2$ for at least the past several hundred years. The ^{14}C content of a homogeneous, steady state C reservoir with turnover times of 10, 50 or 100 years is compared with that of the atmosphere through time.

tracer for C cycling on decadal time-scales. Carbon reservoirs such as SOM that exchange with the atmosphere reflect the rate of exchange through the amount of 'bomb' ^{14}C incorporated (Figure 1). ^{14}C in atmospheric CO_2 is currently decreasing at a rate of about 8‰ per year (Levin & Kromer 1997) because of uptake by the ocean and dilution by burning of ^{14}C-free fossil fuels. The ^{14}C content of a homogeneous C reservoir in any given year since 1963 may be predicted from the turnover time and the known record of atmospheric $^{14}CO_2$. Utilization of bomb-produced ^{14}C as a continuous isotopic label has advantages over other isotopic methods because it can be used in undisturbed ecosystems and can resolve dynamics that operate on annual to decadal time scales.

Soil organic matter is made up of C fractions that cycle on a continuum of time scales ranging from days to millennia. Because of this complexity, ^{14}C measurements of bulk SOM at a single point in time do not yield useful information about the rate of SOM cycling (Trumbore, in press). At the Harvard Forest, we separated SOM into distinct pools with different characteristic

Organic Horizons

Oi   → Whole Sample   → L_L Leaf Litter

Oe+Oa → Hand Picking   ⟨ L_R Recognizeable root litter

  H   Humified Material

Mineral Horizons

A   - → Sieved to → Density Separation   H → Sieved to 80μ → H > 80μ
     < 2mm   Sodium Polytungstate   and hand picked
     @ 2.1 g/cc → M L_R

                                 H < 80μ

Ap or B   - → Sieved to → Density Separation   H
       < 2mm   Sodium Polytungstate
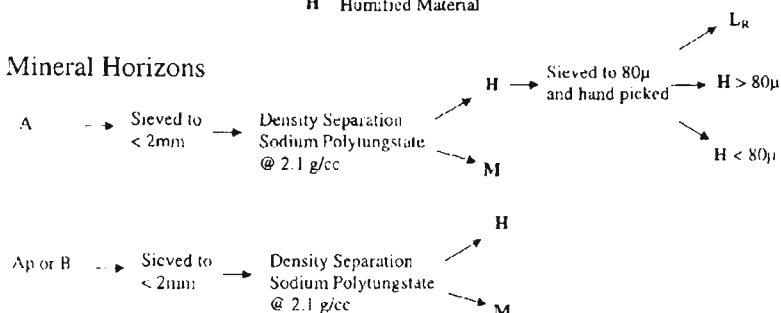   @ 2.1 g/cc → M

Figure 2. Schematic representation of soil sample processing into the homogeneous soil organic matter pools as defined in this paper; L_L or L_R (recognizable leaf or root *litter* respectively), H (undifferentiable SOM which is considered to be microbially altered or *humified*), and M (organic matter associated with *mineral* surfaces). All L_L, L_R and H components are low density (i.e. <2.1 g/cc) while the M components are considered high density (i.e. >2.1 g/cc).

turnover times (see Figure 2): recognizable leaf (L_L) and recognizable fine (<2 mm) root litter (L_R), organic matter that has been transformed by microbial action or humified, but is not stabilized by interactions with mineral surfaces (H), and organic matter that is associated with soil minerals and thus is separable by density (M). These four pools collectively comprise Reservoir-C as defined for this paper and represent detrital C that remains in the soil for one year or more. Carbon pools in SOM that cycle on timescales of less than one year are included in our definition of Recent-C.

Carbon dynamics derived from measurements of ^{14}C in SOM fractions alone tend to underestimate the flux of CO_2 from soils. Heterotrophic respiration is dominated by decomposition of C with short turnover times and small reservoirs that are difficult to measure. The majority of easily measurable SOM stocks represent slowly cycling material with relatively long turnover times. Measurements of ^{14}C in CO_2 can be used to determine the relative contributions of the recalcitrant C, which dominates SOM stocks, and the more rapidly cycling C, which dominates heterotrophic and autotrophic respiration. To derive a below-ground C budget that includes soil respiration, we combined measurements of CO_2 and $^{14}CO_2$ surface fluxes and soil

atmosphere profiles with a model of soil gas diffusion to determine the rate and ^{14}C signature of CO_2 production in soil by horizon. This, combined with the predicted production of CO_2 and $^{14}CO_2$ derived from the L_l, L_R, H and M fractions of SOM, allowed us to partition soil respiration into Recent-C and Reservoir-C.

Site description

The Harvard Forest is a mixed deciduous forest located near the town of Petersham in central Massachusetts. The study area is located on the Prospect Hill Tract (42.54° N, 72.18° W). The terrain is moderately hilly (average elevation 340 m) and currently about 95% forested (Wofsy et al. 1993). The soils are developed on glacial till deposits which are predominantly granitic. Drainage varies from well-drained uplands, which make up most of the area in the flux tower footprint, to very poorly drained swamps. The data reported here are for well drained soils with very low clay content and mapped as Canton Series (coarse-loamy over sandy or sandy skeletal, mixed mesic Typic Dystrochrepts). We sampled soils, soil respiration and soil gas profiles within 100 meters of the eddy flux tower where a multi-year record of soil respiration measured by flux chambers is maintained (Davidson et al. 1998). The sites are within a mixed deciduous stand, dominated by red oak (*Quercus rubra*) and red maple (*Acer rubrum*) with some hemlock (*Tsuga canadensis*) and white pine (*Pinus strobus*). The area comprising our study site was cleared in the mid-1800's, plowed and used primarily for pasture. The pasture was abandoned between 1860 and 1880 (Foster 1992). The regrowing forest was largely leveled by a hurricane in 1938 but has been growing undisturbed since that time.

Methods

Field

We sampled soils using the quantitative pit methodology as discussed by Huntington et al. (1989) and Hamburg (1984). This method involves sampling a large volume of soil to allow calculation of horizon-specific bulk densities. Two 0.5×0.5 m quantitative pits were dug in 1996 to a depth of about 80 cm. Pit locations were selected to be similar to those where Davidson et al. (1998) are monitoring soil respiration and soil CO_2 concentrations and are within 20 meters of their soil respiration collars. In each pit excavation proceeded downward to the base of each pedogenic horizon, which was differentiated

by color and textural changes. In order to minimize sampling errors due to repeated grid placement and removal, the top of each pedogenic horizon was calculated by taking a weighted mean of 25 measurements from within the 0.5×0.5 m grid. This system weights the center nine measurements 4×, the sides of the grid (not including the corners) 2× and the corners 1×. Additional samples which integrated each soil horizon were collected for radiocarbon and total C and N analyses from one of the pit faces. Samples of the forest floor (0.15×0.15 m squares), core samples of A horizons and grab samples of Ap and B horizons were collected in order to analyze the abundance and ^{14}C of roots. During the summer of 1997, a third, shallower (0.17 m × 0.37 m) pit was dug to obtain more data for the O and A horizons. Samples were taken in approximately 2 cm vertical increments to the base of the Ap horizon.

Collars sampled were the same as those used by Davidson et al. (1998) to monitor soil respiration fluxes. Closed dynamic chambers were used for sampling isotopes in soil respiration, as shown in Figure 3. First, atmospheric CO_2 initially inside the chamber cover was removed by circulating air at flow rates of ~0.5 L min^{-1} from the chamber headspace through a column filled with soda lime. Scrubbing continued until the equivalent of two to three chamber volumes had been passed over the soda lime. Then the air flow was switched and flowed through a molecular sieve trap (mesh size 13×). Molecular sieve 13× traps CO_2 quantitatively at room temperatures and then releases it when baked at 475 °C (Bauer et al. 1992). CO_2 was trapped from circulating chamber air until the amount required for isotopic (^{13}C and ^{14}C) measurements (~2 mg of C) was collected. Trapping times varied from about 10 minutes to an hour, depending on the soil CO_2 emission rate. To achieve 100% yields of CO_2 from the molecular sieve traps we have found that it is important to put a desiccant in-line (Drierite) in order to minimize the amount of water getting to the molecular sieve

To measure CO_2 and its ^{14}C signature in the soil atmosphere we collected soil gas samples from stainless steel tubes (3 mm OD) inserted horizontally into soil pit walls (the soils pits were subsequently backfilled). The air within the tubing was first purged by extracting a 15 ml syringe sample through a fitting with a septum. Two more 5 ml samples were then withdrawn from each tube, the syringes were closed with a stopcock, and the CO_2 concentrations of the syringe samples were analyzed the same day in a LiCor infrared gas analyzer as described by Davidson and Trumbore (1995). For the ^{14}C analysis, we filled evacuated stainless steel cans (0.5–2.0 L volume) by attaching them to the buried stainless steel tubes. A flow restrictor was used to fill the cans slowly during a 4 hour period so as to minimize disturbance of the concentration gradient. The soil gas tubes were installed along with
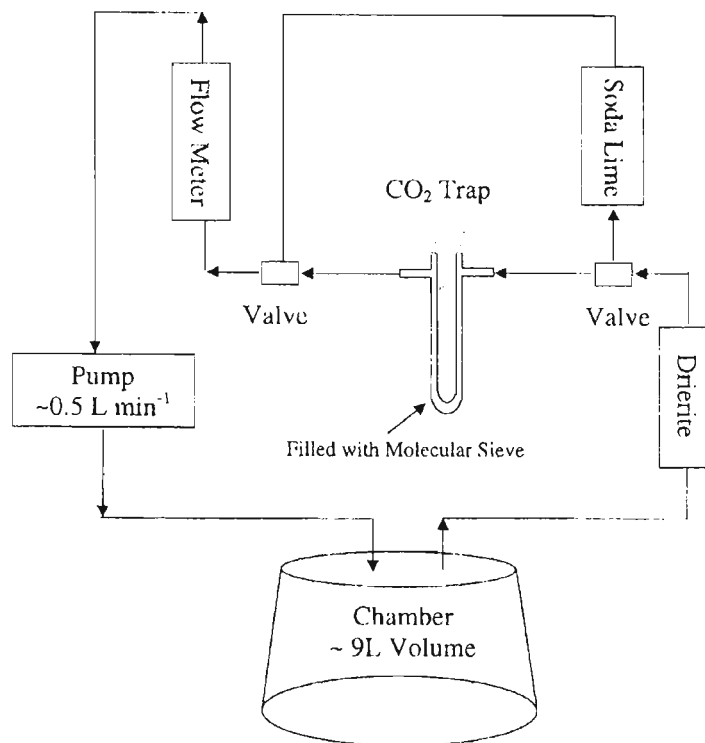
Figure 3. Sampling scheme for trapping CO_2 on molecular sieve (mesh size 13x) using a closed dynamic chamber system. Molecular sieve 13x traps CO_2 quantitatively at room temperatures and then releases it when baked at 475 °C (Bauer et al. 1992). The evolved CO_2 is purified cryogenically.

TDR and temperature probes in one pit in 1995 (not dug quantitatively for bulk density) and in a second pit dug in 1996 (dug quantitatively as discussed above). All pits were located within a few meters of each other. The concentrations of CO_2, water content, and temperature were measured weekly during the summer, once every two weeks during the autumn and spring, and once per month during the winter.

Aboveground litter inputs were collected in six 0.41 by 0.41 m baskets. Each basket was screened at the base and nailed into the ground. Samples were collected, dried and weighed, once in the spring and then on a biweekly basis from late September through early November 1996.

Laboratory

Prior to C and ^{14}C analyses, soil samples were separated into different SOM fractions as defined for this paper (L_L, L_R, H and M) according to procedures outlined in Figure 2. For mineral samples, material that was less than 2.1 g/cc was primarily humified material (H). Fine roots (L_R) were a significant component of low density organic matter only in the A horizon samples. To test for the importance of L_R in determining bulk low density ^{14}C values, after density separation, one A horizon sample was sieved to 80 μ and then hand picked to separate H from L_R components for ^{14}C analysis. The Ap, Bw1 and Bw2 horizons had such a small proportion of fine root material this additional processing was not performed. Once a soil C fraction was isolated, it was split and half the sample was archived while the other half was ground or finely chopped and analyzed for carbon and nitrogen content in a Fisons 5200 Elemental Analyzer. Grinding was done with an air cyclone sampler for the Oi horizon. Oe + Oa samples and root samples were chopped finely with scissors and mineral samples were ground by hand with a mortar and pestle.

In order to quantify fine root biomass, samples were taken by coring or from subsamples dug from our quantitative pits. Samples were frozen immediately after collection, then stored and processed at the Woods Hole Research Center. Oe + Oa horizons were thawed, a sub-sample removed (approximately 8 cm^3) and quantitatively picked for fine roots (<2 mm in diameter). Mineral soils were thawed, sieved through a 5.6 mm sieve and the fine roots that did not pass through the sieve were weighed. In order to pick live versus dead fine roots, a sub-sample of the sieved soil was used (approximately 8 cm^3). Graphite targets of all SOM fractions and soil gas (CO_2) were prepared at UCI using sealed tube zinc reduction methods (Vogel et al. 1992). The ^{14}C analyses of these targets were made by accelerator mass spectrometry (AMS) at the Center for AMS, Lawrence Livermore Laboratory, Livermore, California (Southon et al. 1992). Radiocarbon data are expressed as Δ^{14}C, the per mil deviation from the ^{14}C/^{12}C ratio of oxalic acid standard in 1950, with sample ^{14}C/^{12}C ratio corrected to a δ^{13}C value of -25‰ to account for any mass dependent isotopic fractionation effects (Stuiver & Polach 1977). The precision for radiocarbon analyses prepared using the zinc reduction technique in our laboratory is ±7‰ for values close to modern (0‰).

We measured ^{13}C in a subset of our SOM samples to determine the proper ^{13}C correction for calculating Δ^{14}C values. Low density samples had δ^{13}C values which ranged between -24.78 and -27.57‰. Low density H fractions averaged 0.38‰ higher in ^{13}C than the M fractions (density >2.1 gC m^{-3}). Because the overall variation in δ^{13}C was greater than the difference between

fractions, we used the same correction ($-26‰$) for all SOM. The maximum error introduced to our ^{14}C determination by this assumption ($5.1‰$) is less than the analytical uncertainty of $7‰$.

Measurements of ^{13}C for surface CO_2 flux samples were used to correct for mass dependent fractionation as well as to correct for incomplete stripping of atmospheric CO_2 in the chamber system during CO_2 trapping. The $\delta^{13}C$ value for CO_2 in air ($\delta^{13}C_{atmosphere}$) is $\sim -8.5‰$, whereas the $\delta^{13}C$ of soil respiration should be close to that of SOM ($\delta^{13}C_{soil} = -26‰$). The fraction of air (X) in our sample is then:

$$X = \frac{\delta^{13}C_{measured} - \delta^{13}C_{soil}}{\delta^{13}C_{atmosphere} - \delta^{13}C_{soil}}, \tag{1}$$

and we calculate the $\Delta^{14}C$ of the soil respiration:

$$\Delta^{14}C_{soil} = \frac{\Delta^{14}C_{measured} - X \times \Delta^{14}C_{atmosphere}}{(1 - X)}. \tag{2}$$

The value of $\delta^{13}C_{atmosphere}$ at the level of the respiration collars (~ 5–10 cm) can become as light as $\sim -11‰$ due to atmospheric inversion which traps plant respired CO_2 and any fossil fuel derived CO_2 (particularly in winter) near the surface. Therefore, during each sampling event we trap one air sample and analyze this for $\delta^{13}C$. The resulting $\delta^{13}C$ is then used for $\delta^{13}C_{atmosphere}$ in calculation of equations 1 and 2 for that suite of samples. Values of X ranged between 0.09 and 0.61. The highest values of X are associated with the samples taken in May, 1996, when no attempt was made to strip the initial chamber of atmospheric CO_2 (values in May were 0.61, 0.49, 0.40 and 0.34). For the July, September and December sampling events when 2–3 chamber volumes were stripped prior to sampling, values of X were all below 0.31 with an average of 0.17.

Modeling

Our methods for data analysis involve four modeling components: 1) determination of CO_2 production by horizon, 2) estimation of $\Delta^{14}C$ of CO_2 produced within each horizon, 3) calculation of the amount of CO_2 derived from decomposition of Reservoir-C sources and 4) partitioning of soil respiration into Recent- versus Reservoir-C sources based on a C and ^{14}C mass balance approach. Each modeling component is discussed in turn below.

(1) CO_2 production within each horizon
The production of CO_2 within each horizon was calculated by combining estimates of diffusivity with measured CO_2 concentration gradients. Effective

diffusivity was estimated for each soil horizon using the model of Millington and Quirk (1961), modified for the presence of rocks and for temperature:

$$\frac{D_s}{D_0} = \alpha^{2x} \left(\frac{\alpha}{\varepsilon}\right)^2 \times \frac{(100 - \%RF)}{100} \times \left(\frac{T}{273}\right)^{1.75}, \tag{3}$$

where D_s is the diffusion coefficient in soil, D_0 is the diffusion coefficient of CO_2 in air (0.139 cm^2 s^{-1} at 273 °K at standard pressure), α is the total air-filled porosity, ε is the total porosity, %RF is the percent rock fraction, and T is the soil temperature (°K). As described by Collin and Rasmuson (1988) and by Davidson and Trumbore (1995), the exponential term, 2x, is usually close to 4/3, and can be approximated by the polynomial

$$x = 0.477\alpha^3 - 0.596\alpha^2 + 0.437\alpha + 0.564 \tag{4}$$

The first term in the Millington and Quirk (1961) equation estimates diffusivity in the wet porous soil medium. The second term, which we have added here, adjusts for rock content of these glacial soils, assuming that diffusion of gases through rocks is negligible. The third term, adjusts for the effect of temperature on gaseous diffusion (Hendry et al. 1993). Total porosity is estimated as

$$\varepsilon = 1 - \left(\frac{BD}{PD}\right), \tag{5}$$

where BD is bulk density of the <2 mm soil fraction measured in our quantitatively sampled soil pits, and PD is a weighted average of particle density, assuming that organic matter has a PD of 1.4 g cm^{-3} and soil minerals have a PD of 2.65 g cm^{-3}. Air filled porosity (α) was calculated as the difference between total porosity and volumetric water content measured by time domain reflectometry (TDR) probes, as described by Davidson et al. (1998).

The soil CO_2 concentration profile was fitted to an exponential function (Figure 4):

$$[CO_2]_z = CO_{2\infty}(1 - e^{-\beta z}) + 0.04, \tag{6}$$

where $[CO_2]_z$ is the concentration of CO_2 at depth z in percent, $CO_{2\infty}$ is the fitted asymptotic CO_2 concentration at infinite depth, z is soil depth in cm, β is a fitted parameter, and 0.04 is an adjustment for the approximate concentration of CO_2 at the soil surface (i.e., about 400 μL CO_2 L^{-1} air). The first derivative of this equation is used to estimate the diffusion gradient as a function of depth:

$$\frac{dCO_2}{dz} = CO_{2\infty} \times \beta \times e^{-\beta z}. \tag{7}$$
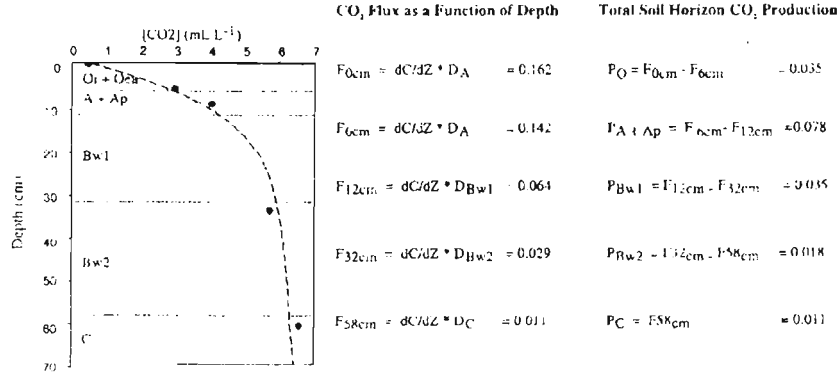
CO₂ Flux and Production Estimates

Figure 4. Calculation of CO_2 flux estimates by depth (F_z, where z indicates the profile depth) and CO_2 production estimates by soil horizon (P_h, where h indicates the specific soil horizon) in gC m^{-2} hr^{-1}. The values shown here are from measurements made on 25 August 1997. Interpolations among similar measurements made throughout the year were summed to obtain annual estimates. These estimates are for well drained soils within the footprint of the eddy flux tower at Harvard Forest.

Applying Fick's first law and combining equations, the flux of CO_2 at a given depth (F_z) can be calculated from the product of the diffusion gradient and the effective diffusivity:

$$F_z = \frac{dCO_2}{dz} \times D_s \times \frac{52700}{T} = CO_{2\infty} \times \beta \times e^{-\beta z} \times \alpha^{2x} \left(\frac{\alpha}{\varepsilon}\right)^2 \times$$

$$\times \frac{(100 - \%RF)}{100} \times \left(\frac{T}{273}\right)^{1.75} \times D_o \times \left(\frac{52700}{T}\right), \qquad (8)$$

where F_z has units of gC m^{-2} hr^{-1}, and where 52700/T is the factor needed to convert to these units.

Using this equation, the flux at the top of each mineral soil horizon (see Figure 4) was calculated for each sampling date in each of the two instrumented soil pits. Our approach to calculating diffusivity differs from many others (e.g., de Jong and Schappert 1972; Johnson et al. 1994; Mattson 1995), in which the flux was calculated from an assumed linear diffusion gradient between two points where CO_2 concentrations were measured. The exponential fit used here for characterizing the CO_2 profiles (Figure 4), while imperfect, appears more appropriate.

Finally, estimation of the production of CO_2 within each genetic horizon (P_h) was calculated from the difference between the flux at the top and bottom of a given soil horizon such that

$$P_h = F_{h-out} - F_{h-in}, \tag{9}$$

where F_{h-out} and F_{h-in} correspond to the appropriate F_z (Figure 4). Production within the O horizon was estimated by the difference between the mean of the six surface chamber flux measurements and the calculated flux at the top of the A horizon. This approach avoids the difficult problem of estimating diffusivity in the O horizons, where small differences in measured bulk density and water content (both of which are difficult to measure well) would have a large effect on our estimate, and where diffusion may not always be the dominant mechanism of gas transport.

(2) $\Delta^{14}C$ of CO_2 produced within each horizon

The total CO_2 and $^{14}CO_2$ flux leaving a soil horizon results from a mixture of the CO_2 that is diffusing through that horizon and that which is produced within the horizon. Therefore, based on horizon specific estimates of CO_2 production (P_h) and measurements of the ^{14}C in CO_2 coming into (ΔF_{h-in}) and going out (ΔF_{h-out}) of a subset of the soil horizons (in this notation, Δ refers to ^{14}C of F in ‰ units, and not "change in F"), we can use a simple mixing equation to calculate the ^{14}C of CO_2 produced within that horizon (ΔP_h) from both Recent- and Reservoir-C sources. The equations used to calculate ΔP_h (in ‰ units) from CO_2 production rates and fluxes (in gC m^{-2} yr^{-1}) are Equation 9 and

$$(F_{h-in} + P_h) \times \Delta F_{h-out} = F_{h-in} \times \Delta F_{h-in} + P_h \times \Delta P_h \tag{10}$$

In this approach chamber measurements of ^{14}C in CO_2 from the surface efflux serve as ΔF_{h-out} for the O horizon and are used to calculate ΔP_h for the entire O horizon. We lumped O, A and Ap horizons (representing the top $\sim$15 cm) because of the large variability in the $^{14}CO_2$ data available for constraining the O/A and A/Ap transitions.

(3) Decomposition of Reservoir-C

We calculate decomposition of Reservoir-C fluxes by first calculating turnover times for each SOM component using its ^{14}C signature and then calculating a decomposition flux based on that turnover time.

(3.1) SOM turnover times from ^{14}C

We used two approaches to determine turnover times for SOM fractions from radiocarbon measurements. For organic matter in the Oi and mineral horizons

(Ap and B), we used a time-dependent, steady state model as presented in Trumbore et al. (1995):

$$C_{(t)} \times R_{som(t)} = I \times R_{atm(t)} + C_{(t-1)} \times R_{som(t-1)} - k \times C_{(t-1)} \times$$
$$\times R_{som(t-1)} - \lambda \times C_{(t-1)} \times R_{som(t-1)}, \qquad (11)$$

collecting terms:

$$R_{som(t)} = \frac{I \times R_{atm(t)} + (C_{(t-1)} \times R_{som(t-1)} \times (1 - k - \lambda))}{C_{(t)}}, \qquad (12)$$

where:

C   = Stock of carbon for the given C pool in gC m^{-2}

I   = Inputs of C above and below ground in gC m^{-2} yr^{-1}

k   = Decomposition rate of SOM in yr^{-1}

R   = $\left(\frac{\Delta^{14}C}{1000} \right) - 1$

R_{atm} = The ratio of ^{14}C in the atmosphere normalized to a standard.

R_{som} = The ratio of ^{14}C in the given SOM pool: L, H or M, normalized to a standard.

λ = radioactive decay constant for ^{14}C = 1/8267 years.

t   = time (year) for which calculation is being performed

I and k are adjusted to match both observed C inventory and ^{14}C content for the fraction in 1996. Note that the R_{som} at any time t, depends not only on the $R_{atm(t)}$ but on both C inventory and R_{atm} of previous years.

For the Oe + Oa and A horizons that have accumulated above the plow layer since abandonment between 1860 and 1880, we used a nonsteady state model that matches both the total amount of C and ^{14}C in 1996. We assumed zero initial C in 1880. Assuming constant I and k, the amount of carbon initially added in each year j (since 1880) that remains and can be measured in 1996 (C_j) will be

$$C_j = I \times e^{-k(1996-j)}, \qquad (13)$$

The ^{14}C signature of C_j will be $R_{atm(j)}$. Therefore the total amount of carbon and radiocarbon measured in 1996 is shown by Equations 14 and 15, respectively:

$$C_{1996} = \sum_{j=1880}^{j=1996} C_j, \qquad (14)$$

$$R_{som(1996)} = \frac{\sum\limits_{j=1880}^{j=1996} R_{atm(j)} \times C_j}{\sum\limits_{j=1880}^{j=1996} C_j}. \tag{15}$$

Again, I and k were adjusted until they matched observations of C and ^{14}C for each fraction in Oe + Oa and A horizons. The rate of accumulation of carbon for a given fraction in 1996 is the difference in C inventory calculated for 1995 and 1996.

Both steady state and nonsteady state accumulation models assume (1) all carbon within a given SOM fraction (L_L, L_R, H, or M) is homogenous with respect to decomposition; (2) the time lag between photosynthetic fixation and addition of fixed C to SOM is one year or less (i.e. the Δ^{14}C of C added to each SOM fraction each year is equal to $R_{atm(j)}$) and (3) radiocarbon does not fractionate during respiration. We have already corrected for mass-dependent fractionation effects when calculating Δ^{14}C values. Any time lag that does exist between photosynthetic fixation and addition of fixed C to SOM (contrary to Assumption 2) will cause an overestimation of turnover time (TT) equal to this lag (Thompson & Randerson 1999). Assumption 2 holds for the majority of aboveground litter inputs (deciduous leaves) which are fixed and fall to the ground within one year. Effects of this assumption with respect to other SOM inputs will be discussed later in the text. Figure 5 shows the Δ^{14}C of a SOM fraction as a function of turnover times in 1996 for both the steady-state and nonsteady state models. Significant differences between approaches appear only for fractions with turnover times greater than about 25 years. This is because the assumption of zero initial carbon in 1880 in the accumulation model limits the amount of pre-bomb ^{14}C in the SOM that is available to dilute the post-bomb carbon that has accumulated since 1963.

(3.2) *Calculating SOM decomposition fluxes*

Decomposition fluxes for the L_L, H and M components of SOM are determined as the inventory in each fraction divided by the turnover time derived from ^{14}C. Since the turnover times for fine roots are too uncertain (as will be discussed in the results section), we treat the flux from L_R as an unknown and solve for it in the C and ^{14}C mass balance section.

The turnover times derived from ^{14}C data may represent the time scales for C loss via several mechanisms, including (1) decomposition loss of CO_2 to the atmosphere; (2) C transfer to another SOM fraction (for example litter to humified material, i.e., L_L or L_R to H or M); or (3) loss by leaching. Data
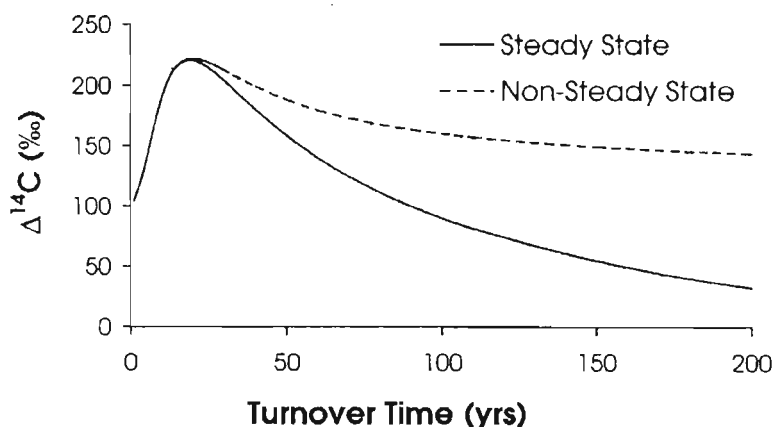
Figure 5. The predicted $\Delta^{14}C$ value in 1996 for homogeneous C reservoirs as a function of turnover times. The curves represent results for our steady state and nonsteady state (accumulation) models.

for dissolved organic carbon (DOC) transport from similar stands in Harvard Forest are available from Currie et al. (1996), and show leaching losses to be minor compared to the other fluxes, except in the O horizon where DOC loss is approximately 20 gC m^{-2} yr^{-1}. This loss is only a few percent of the total annual CO_2 flux, hence we have excluded it from consideration here. Consequently, we assume all loss to be from decomposition or transfer from one C fraction to another.

We model the litter components L_L and L_R as having two fates: decomposition to CO_2 or transfer to the H or M fractions. For the H and M fractions we assume that their source of C is transfer from L_L and L_R fractions and that their most important loss process is decomposition to CO_2. The flux of CO_2 derived from decomposition of leaf litter (F_{LL}) is the inventory of leaf C divided by its turnover time, corrected for the fraction of L_L that is transferred to the H + M pools. Since we cannot independently partition the flux of L_L into either CO_2 or a transfer flux, we bracket our estimates by assuming two extreme cases in which all of the H + M inputs come from either (1) L_L or (2) L_R.

(4) *Partitioning of soil respiration sources*

The total amount of radiocarbon in soil respiration equals the amount of CO_2 derived from Recent-C sources (root respiration and C metabolized within one year of original fixation; R), plus that derived from Reservoir-C (decomposition of L_L, L_R, H and M fractions that reside in the soil for longer than

one year). If the $\Delta^{14}C$ signatures of these components differ significantly, we may use a mass balance approach to determine the relative contribution of each to total soil respiration. We use an isotopic mass balance based on estimates of CO_2 production, the $\Delta^{14}C$ in CO_2 and ^{14}C-derived estimates of decomposition fluxes from the SOM fractions. For the whole soil profile, equations of mass balance for C and ^{14}C are

$$P = F_R + F_{LL} + F_{LR} + F_H + F_M \qquad (16)$$

and

$$P \times \Delta P = F_R \times \Delta R_{atm(1996)} + F_{LL} \times \Delta L_L + F_{LR} \times \Delta L_R +$$
$$+F_H \times \Delta H + F_M \times \Delta M. \qquad (17)$$

In Equations 16 and 17, P is the total annual soil respiration flux and F_R is the flux of CO_2 derived from Recent-C. F_{LL}, F_{LR}, F_H and F_M are fluxes of CO_2 derived from their respective Reservoir-C sources. The Δ values required for the ^{14}C mass balance are either measured (for ΔL_L, L_R, ΔH and ΔM), assumed to equal $\Delta R_{atm(1996)}$ (for Recent-C), or calculated from CO_2 and $^{14}CO_2$ fluxes (ΔP). For the soil profile as a whole, P and ΔP are the measured surface flux and its $\Delta^{14}CO_2$ signature respectively.

We then solved Equations 16 and 17 for the remaining unknowns, F_R and F_{LR}. Since C stocks and rates of C turnover vary vertically within the soil profile, the relative proportions of CO_2 from F_R versus the SOM fractions will vary with soil depth and horizon. Equations 16 and 17 may also be written and solved for each individual soil horizon. However, because of difficulties in characterization of the O/A horizon transition, and uncertainties in the production of roots as a function of depth, we have combined the O + A + Ap horizons and performed the ^{14}C mass balance on only three layers: the O + A + Ap (uppermost 15 cm of soil), B and C horizons.

Results

Carbon inventory

Average carbon stocks are shown in Table 1 by pedogenic horizon. Carbon stocks decrease rapidly with depth at all sites, from 470 gC Kg^{-1} dry soil in the O horizons to less than 10 gC Kg^{-1} dry soil for the Bw2 horizon. We report carbon inventories only to the bottom of the Bw2 horizon because the presence of large boulders limited our ability to measure bulk densities below this depth. The total C stock averages 8.8 kgC m^{-2}, with the majority of C

Table 1. Carbon stocks by soil horizon.

Horizon	Bulk density[1,2] (g cm^{-3})	Soil carbon[3] (gC Kg^{-1} soil)	Bottom depth[4] (cm)	Total C stock[5,6] (gC m^{-2})	Low density SOM			High density SOM
					Leaf litter L_L[7,8] (gC m^{-2})	Fine root detritus L_R[7,8] (gC m^{-2})	Humified H[8,9] (gC m^{-2})	Mineral associated M[8] (gC m^{-2})
Oi	0.06 (0.01)	450 (20)	2 (1)	380 (110)	380 (110)	0	--	NA
Oea	0.1 (0.02)	470 (10)	6 (1)	1640 (750)	–	230 (40)	1410 (750)	NA
A	0.35 (0.03)	270 (30)	10 (2)	2400 (820)	–	60 (25)	1780 (630)	560 (200)
Ap	0.54 (0.13)	60 (1)	16 (2)	2620 (660)	–	70	790 (200)	1760 (450)
Bw1	0.85 (0.07)	20 (1)	32 (4)	1245 (190)	–	4	40 (10)	1200 (180)
Bw2	0.93 (0.04)	6 (1)	59 (3)	510 (110)	--	1 (1)	5 (1)	500 (110)
Total				8800 (1310)	380 (110)	360 (70)	4030 (1000)	4020 (540)

[1] Gravel free bulk density (i.e. less than 2 mm).

[2] For Oi, Oe + Oa and A horizons $n = 3$, standard error in parenthesis; $n = 2$ for all other horizons; range in parenthesis.

[3] For all horizons $n = 2$; range in parenthesis.

[4] For Oi, Oe + Oa and A and Ap horizons $n = 3$, standard error in parenthesis; $n = 2$ for all other horizons; range in parenthesis.

[5] Includes live root mass below the Oi horizon; total error in parenthesis.

[6] C stock calculated using a z value (not shown) that accounts for waviness of horizon boundary and rocks.

[7] On a dry weight basis; error term includes extrapolation from subsample to whole soil where subsample $n = 3$ to 5, otherwise $n = 1$ and no error is shown.

[8] Low density means $\rho < 2.1$ g/cc, high density $\rho > 2.1$ g/cc.

[9] Calculated by subtracting total roots from the total low density SOM.

(80%) in the upper 15 cm, which make up the organic and A + Ap horizons. Measured litterfall inputs to the O horizon were 150 gC m^{-2} yr^{-1} in 1996.

The fraction of soil volume taken up by rocks is spatially variable. In two of the three pits the O and A horizons were much less rocky (0–2% rocks) than the B horizons (10–35% rocks). However, one of our three pits had no less than 15% rocks in all horizons down to 60 cm. Spatial heterogeneity in soil C stocks has been studied in rocky forest soils similar to those found at Harvard Forest. Fernandez et al. (1993) show that between 73 and 455 samples are required to quantify C stocks to within 10% depending on soil depth. Huntington et al. (1988) were able to quantify C stocks to within 20% only after digging 60 0.74 × 0.74 m pits. Therefore in this study instead of quantifying variability within a site we focus on the C dynamics for specific profiles and assume C dynamics will be the same even if the inventory of a given SOM fraction varies spatially for sites with similar drainage.

The four rightmost columns of Table 1 show the inventory of the isolated soil C fractions L_L or L_R, H and M. Carbon in low density fractions decreases rapidly with soil depth, from 100% in O horizons to <1% in B horizons. Low density carbon (L_L + L_R + H) makes up 54% of the total soil carbon stock, but is 87% of the carbon in O + A horizons.

Quantitative picking of roots showed they make up 7–19% ($n = 5$) and 1–4% ($n = 5$) of the dry mass in the Oe + Oa and A horizons, respectively. Assuming roots are 50% C by weight, the fraction of carbon in live and dead roots make up ~14% of the total C stocks in the Oe + Oa horizon, decreasing to ~0.2% in the Bw2 horizon. Our estimate of total fine root mass of 360 gC m^{-2} (live + dead) is lower than that of McClaugherty et al. (1982), who found 525 gC m^{-2} (live + dead) in well-drained mixed hardwood soils at a neighboring study area within the Harvard Forest. Fahey and Hughes (1994), found ~320 and 350 gC m^{-2} (live + dead) in June and October respectively in a mature northern hardwood forest. Our values also decrease more rapidly with depth than those of McClaugherty et al. (1982) who found 70, 55, and 15 gC m^{-2} for 15–30, 30–45 and >45 cm respectively for live + dead fine roots. In addition, our ratio of live:dead fine roots (data not shown) at all depths are significantly greater than those reported by McClaugherty et al. (1982), suggesting either differences in procedures for distinguishing live from dead roots or that we sampled during a seasonal maximum in live root abundance. Technically, it is the dead roots, not the live roots, that are decomposing and contributing to CO_2 fluxes. Therefore, live roots should not be considered part of the SOM. However, because we were not able to reliably distinguish live from dead with confidence, we report them together as L_R.

CO₂ production estimates

Total soil respiration as determined from chamber measurements in 1996 was 840 gC m^{-2} yr^{-1} (Davidson & Savage, unpublished data). Production rates for CO_2 (Figure 4) by soil horizon were 190, 340, 235 and 75 gC m^{-2} yr^{-1} for the O, A + Ap, B and C horizons respectively. The estimates of CO_2 production within each soil horizon include uncertainties associated with the diffusion model, the exponential fit of the CO_2 concentration profiles, and, in particular, measures of rock content. We used the average rock content of two quantitatively sampled soil pits dug in 1996. One had almost no rocks in the O and A horizons while the other had 20–30% coarse fragments. Repeating the calculations assuming either no rocks or the higher estimate of rock content, changed CO_2 production rates for the A horizons by roughly 50 gC m^{-2} yr^{-1}.

Radiocarbon in SOM fractions

The average radiocarbon content and the range of values measured in the isolated SOM fractions are plotted by horizon in Figure 6. $\Delta^{14}C$ values for the low density SOM (L_L or H) fractions increase from the Oi horizon (L_L) where values are 132±8‰ to a maximum in the Oe + Oa horizon (H) of 200±19‰. Humified material in the A horizon is 121‰, and its ^{14}C signature decreases rapidly in the Ap and B horizons. Within all mineral horizons, the low density carbon, which is primarily humus (H), has consistently higher $\Delta^{14}C$ values than mineral-associated (M) carbon, with the largest difference (55‰) in the Ap horizon. Large negative $\Delta^{14}C$ values in both H and M fractions in the Bw1 and Bw2 horizons indicate the majority of soil carbon at these depths has not exchanged with the atmosphere since 1950 and has, in fact, remained in the soil long enough for significant radioactive decay to occur (half life = 5730 years).

The $\Delta^{14}C$ content of live and dead fine roots by horizon are also shown in Figure 6. All roots have $\Delta^{14}C$ values between 134 and 238‰, significantly higher than the atmosphere or live deciduous leaves sampled during 1996 (97±7‰). Live roots on average have lower $\Delta^{14}C$ values than dead roots, and $\Delta^{14}C$ values for both increase with soil depth. The $\Delta^{14}C$ values we measure for live fine roots are surprisingly high, as fine roots in this size class (<2 mm diameter) are thought to have annual or faster turnover based on studies that calculate turnover from the ratio of fine root biomass to production (see review by Vogt et al. 1986) and on direct measurements from rhizotrons, minirhizotrons or root screens (Burke & Dudley 1994; Hendrick & Pregitzer 1992; Hendrick & Pregitzer 1993; Fahey & Hughes 1994). There are three possible interpretations to explain our data. First, the bulk of the fine root mass may live for significantly longer than one year. Second, carbon in roots

still significantly higher than the atmosphere in 1997 (92±7‰), confirming that at least this one live root contained relatively "old" carbon. This one root may not be representative of all species and growth forms, which were averaged during the 1996 quantitative root picking.

The 1997 root data show that different parts of the same root have C that differs in age by 2 years. Thus, if longer turnover is the explanation for elevated ^{14}C, fine roots even 1 mm in diameter and less may not be acting as a single pool with one TT. The data would then imply that the tips (a small part of the mass) may turn over significantly faster than the rest of the root (the bulk of the mass). In a manner analogous to SOM stocks, the most recalcitrant root biomass pool is the largest fraction of the total root biomass pool and is the portion most easily separated from a soil sample for analyses.

Presently, we do not know which of the three above hypotheses for explaining high Δ^{14}C values in root biomass is correct, and additional research is being conducted to address this important issue. We can however, proceed with our mass balance approach without calculating any turnover times for fine roots based on their Δ^{14}C values. We instead solve for F_{LR} as one of the unknowns. However, even without an understanding of the mechanism, our Δ^{14}C data show that live fine roots, upon their death, are adding carbon to SOM that averaged roughly 165‰ in 1996 and that must have been fixed on average 7±1 years previously.

Figure 7 shows the distribution of C and ^{14}C among the different low density components in O and A horizons. In the O horizon (Figure 7(A)), deciduous leaf parts had Δ^{14}C values that increased with depth, from 113‰ in the Oi to 132‰ in the Oe + Oa, where leaves became difficult to recognize. Radiocarbon in fine roots, which represent 14% of O horizon C (Table 1), ranged from 153‰ (live roots) to 188‰ (dead roots). The remaining, undifferentiable, material (H) had a Δ^{14}C value of 201±19‰ (n = 2). Visual inspection shows H, which contained 63% of C in this O horizon sample, consisted of extremely fine root fragments (<0.5 mm), and dark, humified material that could not be identified.

Figure 7(B) shows the distribution of C and Δ^{14}C for the A horizon low density fraction, which was sieved to 80μ and hand picked to remove roots. The measured components range in value from 48‰ to 266‰. The >80 μ material makes up 82% of this low density sample and has a Δ^{14}C of 130‰. Roots (live + dead) are 7% of the carbon with a weighted average Δ^{14}C of 256‰ (n = 2). The <80 μ fraction had the lowest measured ^{14}C value (48‰). The inventory-weighted average Δ^{14}C for all four components was 132‰. To calculate a flux for the H component alone, we take a stock weighted ^{14}C for the two components greater and less than 80μ (Figure 7(B)) which equals 121‰ (0.82*130‰ + 0.09*48‰)/0.91.

A. O Horizon

[153±3‰ (live roots)
188±7‰ (dead roots)]
14%

☐ Undifferentiable (H)

☐ Recognizeable (L_L)
leaves and needles

☒ Root Litter (L_R)

Oe+Oa

Oi

23%
[113±7 - 132±8]

Oe+Oa

63%
[201±19‰]

B. A Horizon

7%
[256‰]

2%
[no data]

9%
[48±7‰]

☐ > 80 μ, Undifferentiable (H)

☐ < 80 μ, Undifferentiable (H)

☒ > 80 μ, Root Litter (L_R)
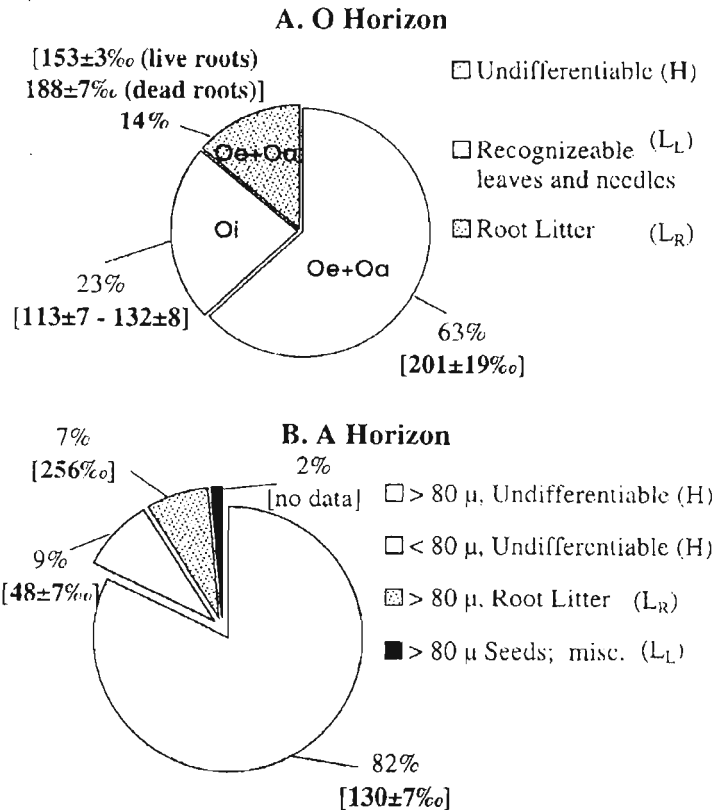
■ > 80 μ Seeds; misc. (L_L)

82%
[130±7‰]

Figure 7. Heterogeneity of the O (top, Figure 7(A)) and A (bottom, Figure 7(B)) horizons. An error of 7‰ indicates analytical error, as $n = 1$. Errors other than 7‰ indicate either a range ($n = 2$) or standard error of the mean ($n = 3$). Values for the O horizon (Figure 7(A)) represent a composite of several samples and are representative of an average O horizon. Values for the A horizon in Figure 7(B) represent the results of quantitative sieving and picking one sample as outlined in Figure 2. The roots in the A horizon represent a stock weighted mean of two samples representing roots of two morphological types with values of 231±7‰ and 266±7‰.

Radiocarbon in the 1996 atmosphere at Harvard Forest

Partitioning of soil respiration using isotopic mass balance requires that we know the $\Delta^{14}C$ of CO_2 for the atmosphere in 1996 (variable $\Delta R_{atm(1996)}$ in Equation 17). For 1996, this value was obtained from two measurements of air within the Prospect Hill Tract on 11 July 1996, and one measurement of

a live deciduous leaf collected on the same date. The three values are 98±7, 96±6 and 97±7‰, averaging 97±1‰. We assume the C lost within a year of being fixed by photosynthesis, including root respiration and decomposition of labile SOM, will have this value in 1996. We support this asssumption with two lines of evidence. First, Horwath et al. (1994) performed a whole tree labeling study on two year old, three meter tall tulip poplar trees. They found that respiration of labeled C from the roots occurred within 12 hours of labeling, the peak activity in respiration was measured after two days and within two weeks the activity of root respiration was less than 5% of the maximum value. Second, three fruiting bodies of the genus *Boletus*, a mycorrhizal fungal symbiont, collected in 1996 at Harvard Forest had $\Delta^{14}C$ of 97, 99 and 98‰. The fact that their $\Delta^{14}C$ signature is the same as the 1996 atmosphere, indicates they are living off of Recent-C substrates, namely root exudates, and not the relatively ^{14}C enriched Reservoir-C of the O horizon in which they are rooted. Since root exudates are Recent-C it follows that maintenance metabolism by trees in this ecosystem must also be respiring Recent-C.

Radiocarbon in soil CO_2

Figure 8(A) and (B) shows CO_2 fluxes and measured $\Delta^{14}C$ in CO_2 of the surface flux for four sampling periods in 1996. The CO_2 fluxes shown in Figure 8(A) range between 40 and 200 mgC m^{-2} hr^{-1}. The largest values measured were in the early summer and the lowest in the winter (see Davidson et al. (1998) for more complete seasonal CO_2 flux data). All flux measurements were made within 1-2 days of ^{14}C sampling, except for the late September sampling event when fluxes were measured 8 days previously. Measured $\Delta^{14}C$ in CO_2 values for 1996 in Figure 8(B) range from 103-176‰ and are all higher than the atmospheric $\Delta^{14}CO_2$ for 1996 (97±1‰). Hence, decomposition of organic matter in the L_L, L_R and H fractions, primarily in the O and A horizons, and which have $\Delta^{14}C$ values greater than 100‰, must contribute significantly to the total soil CO_2 flux. In 1996, the highest $\Delta^{14}C$ values in soil respiration were observed in the spring and summer with values of 138 and 149‰ respectively. The lowest values were in fall and winter where $\Delta^{14}C$ drops to 111 to 121‰ respectively. The data from Figure 8(A) and (B) were used to calculate an annual flux weighted mean $\Delta^{14}C$ in soil respiration for 1996 of 128±9‰ (*n* = 11).

The concentration-weighted annual average $\Delta^{14}C$ in CO_2 for soil air (by depth) is shown in Figure 6 and Table 2. $\Delta^{14}C$ values range from 128 to 136‰, all significantly greater than the 1996 atmosphere (97±1‰). Temporal variability in measured $\Delta^{14}C$ values are greatest at 10 cm (the A horizon) where the values used for averaging ranged from 113 to 161‰. At all other

Table 2. Calculation of high and low density decomposition fluxes with associated $\Delta^{14}CO_2$ and measured profile $\Delta^{14}CO_2$.

Horizon	Stock (g C m^{-2})	Δ^{14}C of SOM[1] (‰)	Turnover time[2] (years)	CO_2 flux (g C m^{-2}-yr^{-1})	Flux horizon total (g C m^{-2}-yr^{-1})	Δ^{14}C of SOM horizon total (‰)	Measured profile $^{14}CO_2$[3] (‰)
Oi (L$_L$)	380	132 (8)	4	25–95[4]	85–160	146	136
Oe + Oa (L$_R$)	230	188[5]	NA	NA			
Oe + Oa (H)	1410	201 (19)[6]	40	35	35		
A (L$_R$)	60	216[5]	NA	NA	<22	115	130
A (H)	1780	121[7]	73 to >100[8]	<18			
A (M)	560	68 (26)	130 to>200[8]	<4			
Ap (H)	790	24.5 (28)	220	4	8	15	–
Ap (M)	1760	−31(20)	480	4			
Bw1 (H)	40	−97 (18)	990	0.04	<2	−118	131
Bw1 (M)	1200	−119 (17)	1200	1			
Bw2 (H)	5	−129 (40)	1300	0.004	<1	−171	128
Bw2 (M)	500	−172 (4)	1760	0.3			

NA = Not applicable, see text for details.

– = no data.

[1] Values are the average for two pits with range in parenthesis.

[2] A nonsteady state model is used for the Oe + Oa and A horizons and a steady state model used for Oi, Ap and deeper horizons.

[3] Represents an annual concentration weighted average of the measured Δ^{14}C in CO_2 at the boundary with the horizon below.

[4] Represents a range based on assuming all loss is as CO_2 or that 100% of inputs to the H + M fractions are from leaf litter.

[5] Represents the Δ^{14}C samples picked for dead roots (n = 1).

[6] Represents the humified organic material after quantitative root picking for the Oe + Oa (n = 2).

[7] Represents a weighted Δ^{14}C value for the two humified (H) components shown in Figure 7(B), also text for discussion.

[8] Range reflects that the Δ^{14}C values may include pre-disturbance C. The minimum represents the steady state case.
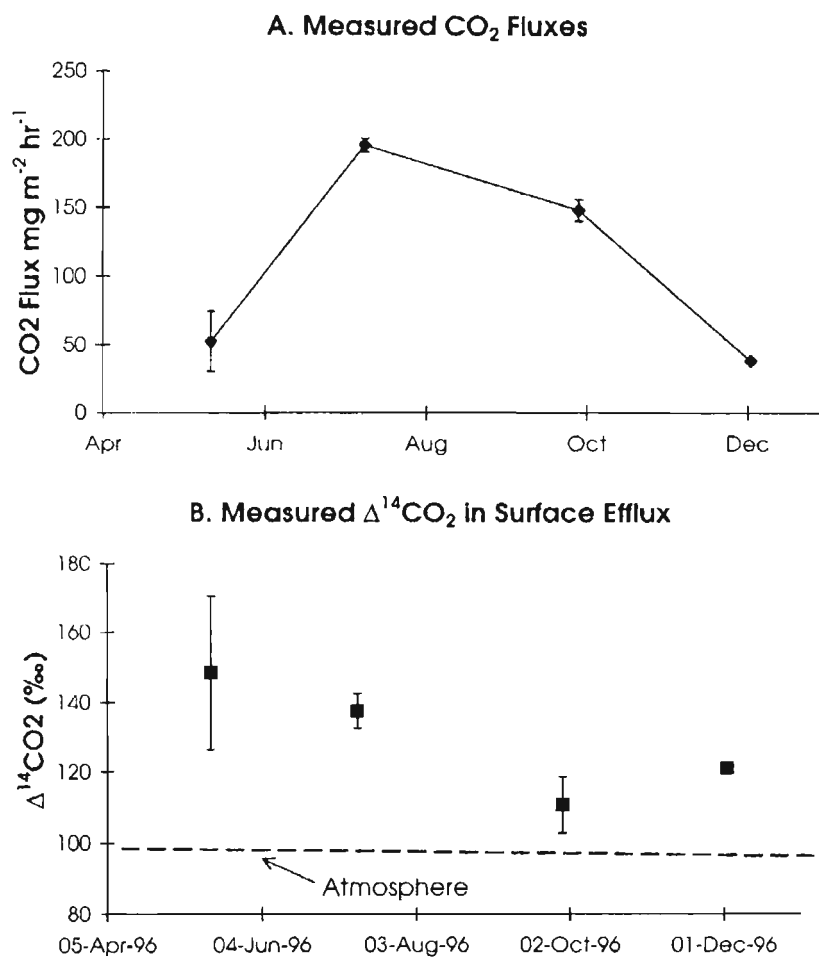
A. Measured CO_2 Fluxes

B. Measured $\Delta^{14}CO_2$ in Surface Efflux

Figure 8. CO_2 fluxes for 1996 (top, Figure 8(A)). Error bars represent standard error ($n = 6$). $\Delta^{14}CO_2$ of soil CO_2 efflux (bottom, Figure 8(B)) Error bars where present represent standard error ($n = 3$) except in December 1996 where $n = 1$.

depths the annual variation was 20‰ or less. The fact that the $\Delta^{14}C$ in CO_2 is greater than either atmospheric CO_2 or H and M carbon in the mineral Ap and B horizons shows decomposition fluxes must be dominated by root litter (F_{LR}) which has much higher $\Delta^{14}C$ values.

Fluxes and turnover time of C in SOM fractions

Table 2 shows the turnover times for SOM fractions as derived from their radiocarbon content and the functions shown in Figure 5. Modern $\Delta^{14}C$

values ($>100‰$) have two possible turnover times for each $\Delta^{14}C$ value. For example, the $\Delta^{14}C$ of H in the Oe + Oa horizon is 201‰ which corresponds to turnover times of either 9 or 40 years (Figure 5, nonsteady state model). Based on the requirements for total CO_2 and $^{14}CO_2$ production in each horizon, we chose the longer turnover time for the H fraction in this and other horizons. We did not calculate turnover times for root litter (L_R) because of the potential for a significant lag time to affect the $\Delta^{14}C$ values measured in 1996. The time lag would also affect the TT for undifferentiable (H) material that is derived from both leaf litter and fine roots. In each case, failure to correct for any lag will cause overestimation of turnover times by as much as the inferred ^{14}C-derived lifetime of live roots (7 ± 1 years). Failure to account for time lags if roots are the principal source for more recalcitrant organic matter would result in turnover times for H and M fractions that are ~7 years too long.

The flux of CO_2 derived from decomposition of L_L, H and M fractions is calculated in Table 2 from the C inventory and turnover time. Again, no flux is calculated for fine root decomposition (F_{LR}); instead we use the CO_2 and $^{14}CO_2$ mass balance to calculate this below.

Partitioning of soil respiration

Equations 16 and 17 contain three unknowns: F_{LL}, F_{LR} and F_R. We therefore introduce an additional constraint so that we may solve for all three fluxes. As we have defined them, L_L and L_R pools represent the detrital root and leaf material that take longer than one year to decompose and are identifiable in SOM. From the inventory of detrital leaf litter (L_L; 380 gC m^{-2}) measured in the soil, and its turnover time (4 years; Table 2), we calculate the annual flux of C into the L_L pool as 95 gC m^{-2} yr^{-1}. The fate of leaf detritus is either to decompose directly to CO_2 (this is the flux F_{LL}) or to be incorporated into soil humus and mineral pools (H + M). We do not know this partitioning; however, based on C and ^{14}C inventory and our nonsteady state model, we assume the the annual rate of input to the H + M pools equals the decomposition flux from these pools (~70 gC m^{-2} yr^{-1}, see Table 2). We then consider the two extreme cases, where all humus and mineral C is derived from leaf detritus, or all of it is derived from root detritus. F_{LL} is thus constrained to be between 25 and 95 gC m^{-2} yr^{-1}. In Table 3, we use these minimum and maximum values for F_{LL} and solve for the two remaining fluxes, F_{LR} and F_R. Table 3 also shows cases for using the minimum and maximum values for observed $\Delta^{14}C$ of detrital leaf (113 and 132‰) and root (180 and 214‰) pools. The ranges and means of this approach are shown in Figure 9.

Our results from partitioning soil respiration for the entire soil profile using CO_2 and $^{14}CO_2$ mass balance are summarized in Table 3 and Figure 9. Approximately 41% (34%–51%) of CO_2 produced annually is derived from

Table 3. Parameters used and results of respiration partitioning for the whole soil profile based on eight scenarios.

Case	Parameters	Leaf litter F_{LL}^1 (gC m^{-2} yr^{-1})	Leaf litter ΔL_L^2 (‰)	Fine root litter F_{LR} (gC m^{-2} yr^{-1})	Fine root litter ΔL_R^3 (‰)	Recent-C F_R (gC m^{-2} yr^{-1})	Reservoir-C $F_{LL} + F_{LR} + F_{H+M}$ (gC m^{-2} yr^{-1})	Recent-C fraction	Reservoir-C fraction
Case 1	Min L_L. min ΔL_L. min ΔL_R	25	113	**277**	180	**470**	**370**	**0.56**	**0.44**
Case 2	Min L_L. min ΔL_L. max ΔL_R	25	113	**197**	214	**550**	**290**	**0.66**	**0.34**
Case 3	Min L_L. max ΔL_L. min ΔL_R	25	132	**272**	180	**475**	**365**	**0.57**	**0.43**
Case 4	Min L_L. max ΔL_L. max ΔL_R	25	132	**193**	214	**554**	**286**	**0.66**	**0.34**
Case 5	Max L_L. min ΔL_L. min ΔL_R	95	113	**264**	180	**413**	**427**	**0.49**	**0.51**
Case 6	Max L_L. min ΔL_L. max ΔL_R	95	113	**187**	214	**490**	**350**	**0.58**	**0.42**
Case 7	Max L_L. max ΔL_L. min ΔL_R	95	132	**242**	180	**435**	**405**	**0.52**	**0.48**
Case 8	Max L_L. max ΔL_L. max ΔL_R	95	132	**172**	214	**505**	**335**	**0.60**	**0.40**
Average		60		**219**		**493**	**347**	**0.59**	**0.41**
Minimum				**172**		**413**	**286**	**0.49**	**0.34**
Maximum				**277**		**554**	**427**	**0.66**	**0.51**

For all scenarios, $\Delta^{14}C$ of the atmosphere (ΔR) = 97‰, the $\Delta^{14}C$ of total soil respiration (ΔP) = 128‰, the flux of H + M is 70 gC m^{-2} yr^{-1} with a $\Delta^{14}C$ of 135‰. We combine the fluxes and their associated ^{14}C values of the H and M pools because their combined fluxes are relatively low (less than 10% of the annual total). Nonbold face values are parameters used in Equations 16 and 17, while bold faced values are the resulting calculations.

[1] Range reflects the two cases where either none or all of the inputs to H + M fractions are derived from leaf litter.

[2] Range is for the lowest and highest measured values of recognizeable leaf parts.

[3] Range reflects a mass weighted ^{14}C average of all dead roots (180‰) and the highest measured dead root value (214‰).
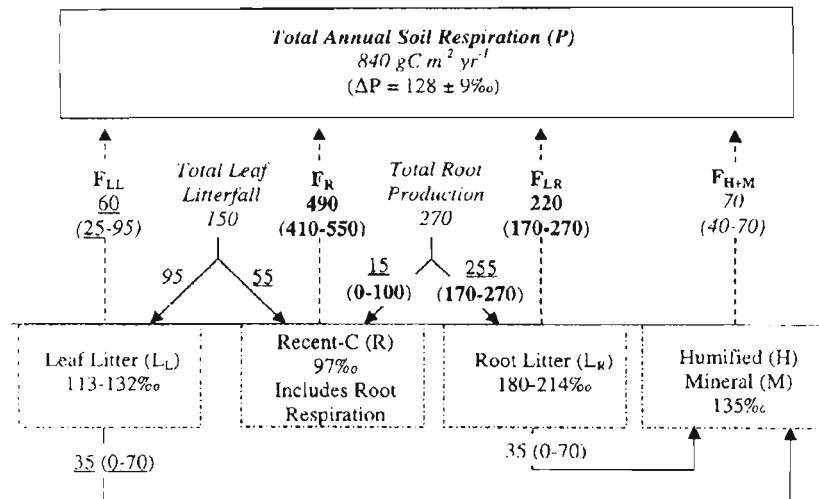
Figure 9. Results of isotopic mass balance approach to partitioning soil respiration into Recent- versus Reservoir-C sources. Solid arrows represent fluxes of organic C, while dashed arrows represent fluxes of CO_2. All units are in $gC\ m^{-2}\ yr^{-1}$ with the average (and range). Production of litter (leaf and root) is assumed to have the isotopic composition of the atmosphere (97‰) in 1996. Bold numbers represent direct results from isotope mass balance model. Italicized numbers are independent measurements or calculated values used to constrain the model (see text for details) and underlined numbers are the resultant fluxes and transfers due to the model results and its constraints.

decomposition of low density SOM with TT greater than one year (L_L, L_R, H and M, i.e., Reservoir-C). The decomposition of H and M fractions with turnover times >40 years contributes only 8% of the total annual respiration flux, with the remaining 33% (26%–43%) from root and leaf litter decomposition (with plant + soil residence times of 2 to 10 years). The fluxes into and out of the H + M pools are constrained by their ^{14}C-derived TT and reservoir size (Table 2) and represents an average over decadal time scales. The average flux of 70 $gC\ m^{-2}\ yr^{-1}$ shown in Figure 9 implies the H + M pool to be in steady state. The uncertainty of 40–70 $gC\ m^{-2}\ yr^{-1}$ is an estimate based on accumulation rates discussed in the following section.

Note that the flux of C into and out of the L_L pool is less than the total annual measured leaf litterfall (150 $gC\ m^{-2}\ yr^{-1}$; Figure 9). We infer that 55 $gC\ m^{-2}\ yr^{-1}$ (1/3rd) of the freshly deposited litter is decomposed in <1 year (and hence is not detected from leaf detritus collected the following summer). Similarly, comparing the range of fluxes of C out of the L_R pool (170–270 $gC\ m^{-2}\ yr^{-1}$, which is equal to F_{LR} + the flux of root C transformed to humus) with annual root production estimates by McClaugherty et al. (1982) of 270

Table 4. Summary of respiration partitioning results.

Horizon	Total respiration $(gC\ m^{-2})$	Fraction total respiration	Fraction produced that is Reservoir-C	Min	Max
Whole Soil	840	1.00	0.41	0.34	0.51
O + A + Ap	530	0.63	0.44	0.35	0.54
B	235	0.28	0.39	0.32	0.45
C	75	0.09	0.37	0.31	0.43

$gC\ m^{-2}\ yr^{-1}$ indicates that 0–100 $gC\ m^{-2}\ yr^{-1}$ of root litter is decomposed in less than one year.

Depth-dependence

Figure 4 and Table 4 show that 310 $gC\ m^{-2}\ yr^{-1}$ or 37% of the total annual soil respiration is produced below 15 cm within the B and C horizons. Comparison of isotopic data for the SOM fractions in these horizons in Figure 6 and Table 2 clearly demonstrates that the two main sources must be root decomposition and Recent-C, because the decomposition of H + M reservoirs accounts for <2 $gC\ m^{-2}\ yr^{-1}$ (with F_{LL} = 0). Application of C and ^{14}C mass balance to the B and C horizons shows that 39% and 37% respectively, of the respiration comes from Reservoir-C (Table 4) and that essentially all of this flux is decomposition of roots with high $\Delta^{14}C$ values relative to the atmosphere. In the upper 15 cm of the soil profile (O + A + Ap) we estimate that 44% is Reservoir-C, with 32% from the decay of leaves and roots with TT of 2–10 years, and 12% from H and M fractions with TT >40 years. More frequent measurements of ^{14}C in total soil respiration and within the vertical profile will allow for more detailed calculation of depth dependence of the make-up of soil respiration. Our measurement of root biomass in the B horizon (Table 1) is insufficient to support the approximately 90 $gC\ m^{-2}\ yr^{-1}$ of decomposition required by the mass balance approach, if fine root mass is homogenous with respect to turnover.

Discussion

Rate of carbon accumulation in SOM

The O and A horizons have accumulated 4.4 $kgC\ m^{-2}$ above the plow layer (Ap horizon) since the late-1800s. This is roughly half the carbon in the soil profile. Carbon pools with turnover times that are less than several decades

(leaf and root litter), which make up 15% of the soil carbon in these horizons, must have achieved steady state with vegetation inputs by 1996. Most of the C in the O and A horizons, however, is in the form of altered, humified (H) material not associated with minerals. The rate of turnover of these fractions is slow enough (40 to 100+ years) that the annual C inputs (I in Equations 11–13) required to support the inventory and ^{14}C observed in 1996 are small (20–50 gC m^{-2} yr^{-1} in the Oe + Oa and 10–30 gC m^{-2} yr^{-1} in the A horizon. The rate of C accumulation in 1996 estimated using our accumulation model is 2–7 gC m^{-2} yr^{1} for the Oe + Oa and 8–23 gC m^{-2} yr^{-1} for the A horizon. The ranges reported bracket the values obtained for different model runs representing mean, low and high values (i.e. ±1 standard deviation) of both C stocks (Table 1) and their ^{14}C values (Table 2). Also included in the range are runs done using the specific C inventories and Δ^{14}C values for each of the two pits (data not shown). Variability in rock content between the two pits affected the overall C stock calculated for each pit and thus the pit with the most rocks had the smallest rates of C accumulation.

While these rates are large compared to storage rates in soils over longer timescales (e.g. Schlesinger 1990; Harden et al. 1992), they are less than the annual net C uptake measured for this ecosystem of ~200 gC m^{-2} yr^{-1} (Goulden et al. 1996). Overall C accumulation rates by the well drained soils which dominate the area within the tower footprint account for 5–15% of this net ecosystem uptake. The predominant wind directions at the tower site are southwesterly and northwesterly. A small area of poorly drained soils close to the tower to the southwest and a swamp 500 m from the tower to the northwest could be larger sinks per unit area than are the well drained soils.

We have assumed the leaf and root litter pools, which have TTs < 10 years, are at steady state. However, if net primary productivity has been increasing as a result of CO_2 or N fertilization, then leaf and root litter pools may be sequestering C. As discussed earlier, annual inputs to L_L and L_R pools are 150 and 270 gC m^{-2} yr^{-1} (see Figure 9). Assuming a 1% per year increase in NPP between 1991 and 1996 and correcting for the inputs respired during the same year then the L_L and L_R pools could also be storing a combined ~20 gC m^{-2} yr^{-1} ((270 − 15) + (150 − 55)) gC m^{-2} yr^{-1}*0.01 yr^{-1}*5 yr) through this period. Combined with the accumulation in the humic fractions of the O and A horizons this could account for as much as 25% of the net ecosystem uptake.

Partitioning of soil respiration

We estimate from our radiocarbon studies that 59% of the soil respiration was derived from C that resided in the plant + soil system for less than one year. Using trenched plots to exclude roots, Bowden et al. (1993) estimate that

root respiration (plus decomposition of roots killed during trenching) was 33% of the total soil respiration at a nearby study site at the Harvard Forest. Using litterfall exclusion and addition manipulations, these authors estimate that 11% of the total soil respiration was from above-ground litterfall less than one year old. Hence, Bowden et al. (1993) estimated a total of 44% of the respiration was derived from C with a residence time in the *soil* system of less than one year. Both radiocarbon measurements and root and litter manipulations have uncertainties, and the best interpretation is probably that these two very different approaches yield estimates that about 50%±10% of the soil respiration is derived from C that is less than one year old. Bowden et al. (1993) also estimate that 30% of soil respiration was from root litter that had resided in the soil more than one year, which is consistent with our radiocarbon data that show somewhat surprisingly long mean residence times for live and dead roots.

Time lags in the soil C reservoir – potential for interannual variability

The measured $\Delta^{14}C$ of total soil respiration is 128±9‰ for 1996 which corresponds to a mean residence time for C in the plant + soil system of 4±1 years. This represents the time an average C atom spends in the plant + soil system since original photosynthetic fixation and includes both root respiration and all decomposition sources. We also calculate the average value for $\Delta^{14}C$ of heterotrophic respiration is 167‰ which corresponds to an average age of 8±1 years. Thus a significant time lag exists between initial C fixation and ultimate respiration by heterotrophs. Therefore, variations in C storage or loss in any one year must partially reflect the net ecosystem uptake of previous years. (Schimel et al. 1997; Fung et al. 1997).

The age of C respired from soil can be used to predict the ^{13}C isotope disequilibrium for Harvard Forest. The ^{13}C isotope disequilibrium is the difference between the ^{13}C signature of atmospheric CO_2 being fixed by plants and the ^{13}C respired from soils. A difference is expected because the ^{13}C in the atmosphere has been decreasing with time due to the addition of ^{13}C-depleted fossil fuels to the atmosphere (e.g. Ciais et al. 1995; Fung et al. 1997). Using the $\delta^{13}C$ trend of −0.02‰ per year (Fung et al. 1997), and an average age of 8±1 years for heterotrophic respiration, we estimate the ^{13}C isotope disequilibrium at the Harvard Forest to be −0.16±0.02‰. This is less than that predicted using the CASA model for temperate deciduous forests (Thompson & Randerson 1999; Fung et al. 1997), largely due to the influence of woody debris in the CASA model. Woody debris is relatively small at the Harvard Forest floor, because of clearing of downed wood following the 1938 hurricane. However, Johnson and Todd (1998) note that woody debris does not seem to be an important source of humified SOM in a deciduous forest

at Oak Ridge, TN. Further work should place more emphasis on measuring radiocarbon in CO_2 respired from decomposing logs to asses the importance of this component to total soil respiration.

Heterogeneity issues

Comparison of the bulk SOM $\Delta^{14}C$ with the $\Delta^{14}C$ in CO_2 at depth (Figure 6) clearly demonstrates that the $\Delta^{14}C$ signature of the SOM alone is not enough to estimate C dynamics. Even with density separations into low and high density pools, $\Delta^{14}C$ of SOM is usually biased toward recalcitrant C stocks. This is particularly true in the mineral horizons where the vast majority of C stocks are hundreds of years old and have large negative $\Delta^{14}C$ values. The small pools of fast cycling SOM (fine roots) with significant amounts of 'bomb C' are effectively diluted beyond isotopic recognition. Our technique of respiration partitioning, which accounts for decomposition via $^{14}CO_2$ measurements, is particularly robust in the mineral horizons where the respiration sources are so isotopically different and have less spatial and temporal heterogeneity.

Methods of estimating bulk soil turnover rates by taking soil C stock divided by CO_2 flux also do not account for soil profile heterogeneity. Particularly in temperate forest soils with significant O and A horizon carbon stocks, this approach will yield poor estimates of the response time of soils to climate change scenarios. Figure 10 shows differences in soil C increase in response to a 10% increase in C inputs for a one-pool model with a turnover time of 25 years versus a multi-pool model representing the well drained soil at Harvard Forest. The one pool model overestimates the amount of C sequestered both in the short and long term. After 100 years, the one-pool model over predicts C storage by almost 600 gC m^{-2}.

Similarly, bulk SOM radiocarbon measurements may also cause an overestimation of decadal scale SOM response. For example, had we not partitioned the low-density carbon in the A horizon into different components (fine roots, undifferentiable material $>80~\mu$ and $<80~\mu$), the bulk ^{14}C value would have been 132‰ with a TT of 66 years. Instead, Figure 7B shows the sample to be composed of components with ^{14}C-derived TTs ranging from ~ 8 to >100 years. Analogous to taking an average TT for the whole soil, the response of SOM would be overestimated if the $\Delta^{14}C$ signatures of the low-density C sample components were represented using the bulk radiocarbon value (see Figure 10).

The turnover times we derive from modeling the ^{14}C content of SOM fractions are averages over several years to decades. Actual decomposition rates in any given year may deviate from these averages. For example soil respiration measured in Harvard Forest well-drained soils in 1997 (a dry year

by estimated heterotrophic respiration are not good predictors of the response time of soils because soil organic matter (SOM) is not homogeneous.

- Measurements of $\Delta^{14}C$ in CO_2 are required to correctly model the C that is actually respiring and to fully understand below ground C dynamics.

- Interpretation of ^{14}C data in SOM at Harvard Forest are complicated by fine root inputs with ^{14}C elevated by $\sim$65‰ relative to the atmosphere, implying that the fine root C was fixed on average 7 ± 1 years ago. We do not currently understand the mechanism behind this lag in radiocarbon input.

- We estimate 41% of total soil respiration comes from decomposition of SOM that decomposes on timescales of 1–100+ years. Of this, 80% involves direct decomposition of leaf and root litter with TT of 2–10 years and 20% represents low density humified C pools and C associated with minerals (H and M respectively) which have TTs on the order of several decades or greater.

- About two-thirds of total soil respiration is produced within the O and A horizons. These organic rich horizons are comprised of (1) small pools of live roots and recent leaf and root litter that have residence times in the plant + soil system of $\sim$1–10 years and (2) relatively large pools of humified root and leaf litter which reside in the plant + soil system for 40–100+ years.

- Radiocarbon measurements of total below ground respiration measure the average time C spends in the plant + soil system from original photosynthetic fixation until respiration by autotrophs or heterotrophs. We estimate this time to be 4 ± 1 years for total soil respiration and 8 ± 1 years for heterotrophic respiration in well-drained soils at Harvard Forest, MA.
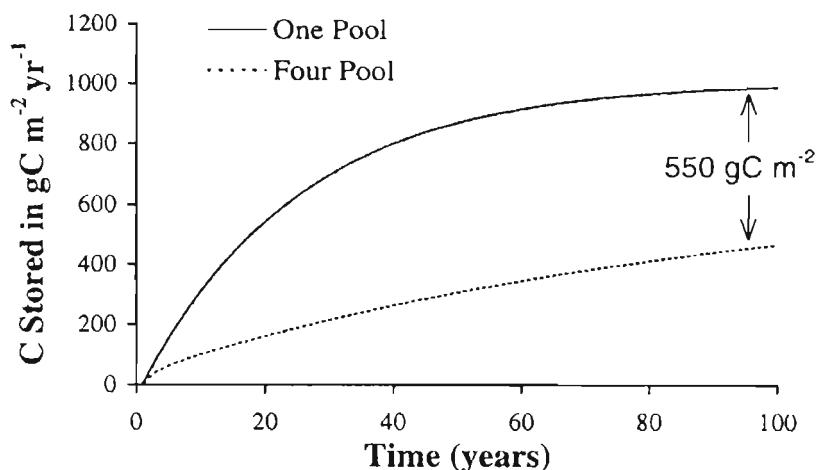
Figure 10. Increase in C storage associated with a 10% increase in NPP for two non-steady state models based on a one pool or four pool representation of soil organic matter stocks. In the one pool case the TT is 25 years and is equivalent to the total soil carbon stocks (8800 gC m^{-2}) divided by the total soil respiration (840 gC m^{-2} yr^{-1}) multiplied by the amount of decomposition from Reservoir-C (41%). In the four pool case stocks and TTs are modeled after those for the Harvard Forest well drained soils with TTs of 1, 4, 80 and 500 years. Both systems are not at steady state; the increases are relative to a nonsteady state run for each case.

relative to 1996) was 660 gC m^{-2} yr^{-1} (Davidson & Savage, unpublished data), compared to 840 gC m^{-2} yr^{-1} for 1996. Monitoring of soil respiration isotopic composition should shed light on whether the reduction in soil respiration in future years is caused by changes in decomposition, changes in root respiration, or both. The power of coupling this approach to measuring soil profiles of CO_2 and $^{14}CO_2$ will allow determination of where in the soil profile changes occur in response to climate.

Conclusions

- SOM pools are accumulating C in well-drained Harvard Forest soils as they recover from historic disturbance. However, the rates of accumulation we infer, 10–30 gC m^{-2} yr^{-1}, are only 5–15% of the 200 gC m^{-2} yr^{-1} observed by the eddy flux tower. More poorly drained soils also in the tower footprint may be accumulating larger amounts of C per square meter area, although they are far more limited in areal extent.
- Measurements of ^{14}C in soil organic matter emphasize organic matter fractions with longer turnover times (TT) which dominate soil carbon inventory. Calculations of TT derived by dividing total C inventory

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