Land cover change – Verhalten von Quantität und Qualität des Bodenkohlenstoffes in sich wandelnden Landschaftssystemen

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“Geography is a living, breathing subject,
constantly adapting itself to change. It is dynamic and relevant.
For me geography is a great adventure with a purpose.”

Michael Palin, Präsident (2009-2012) der Royal Geographical Society

ZUSAMMENFASSUNG

Böden stellen im aktiven Kohlenstoffkreislauf abgesehen von den mittleren und tiefen Wassermassen des Ozeans das grösste Kohlenstoffreservoir dar (Ciais et al., 2013; Houghton, 2014). Je nach Datenquelle und untersuchter Bodentiefe variiert der Kohlenstoffvorrat im Boden zwischen 1500 und 2400 GtC (Batjes, 1996; Jobbágy and Jackson, 2000; Prentice et al., 2001). Waldökosysteme nehmen rund ein Drittel der terrestrischen Fläche ein und speichern in der Biomasse (448 GtC) und in den Böden (747 GtC) mehr als 50 % des globalen terrestrischen Kohlenstoffes (2349 GtC) (Prentice et al., 2001; FAO, 2010; Ussiri and Lal, 2017a). Der restliche Kohlenstoff ist vorwiegend in den Böden (952 GtC; 41 %) von Savannen- und Graslandschaften, Buschökosystemen, Feucht- und Ackerbaugebieten sowie der Tundra gespeichert (Houghton, 2014). In den ersten drei Metern der zirkumpolaren Permafrostböden sind weitere 1000 GtC gespeichert (Tarnocai et al., 2009; Hugelius et al., 2014). Weil das Kohlenstoffreservoir der Böden mehr als zweimal so hoch ist wie jenes der Atmosphäre (590 GtC), haben geringe Zunahmen des Bodenkohlenstoffvorrates grosse Auswirkungen auf den Bodenkohlenstoffvorrat in der Atmosphäre (Lal, 2004; Smith, 2004).

Die Verlinkung des passiven mit dem aktiven Kohlenstoffkreislauf durch die Verbrennung von fossilen Brennstoffen und die Veränderung des aktiven Kohlenstoffkreislaufs durch die Landnutzungsänderungen infolge der steigenden anthropogenen Nachfrage nach Ressourcen haben zu einer Freisetzung von Kohlenstoffverbindungen (v.a. CO_2 und CH_4) während den vergangenen 250 Jahren geführt (Ramankutty et al., 2006, 2008; Ciais et al., 2013). Dabei sind die freigesetzten 600 GtC von den Kohlenstoffsenken „Ozean", „Terrestrisches System" und „Atmosphäre" zu 175, 165 und 260 GtC aufgenommen worden (Ciais et al., 2013). Daher hat sich die Kohlenstoffkonzentration in der Atmosphäre zwischen 1750 und 2015 von 130 auf 400 ppm erhöht (Le Quéré et al., 2016), was eine Verstärkung des natürlichen Treibhauseffektes zur Folge hat. Die globale Durchschnittstemperatur ist seit 1880 um 0.96 C angestiegen und bis 2100 wird mit einer weiteren Zunahme von 1.4 bis 5.8 C gerechnet (Field et al., 2013; Dahlman, 2017).

Das globale Kohlenstoffbudget zeigt eine jährliche Kohlenstoffspeicherung im terrestrischen System von ungefähr 3.1 GtC a^{-1} auf (Le Quéré et al., 2016). Der Anteil, der in Waldökosystemen gespeichert wird, beträgt dabei 2.4 GtC a^{-1} (Pan et al., 2011), was durch die hohe Nettoprimärproduktion zu erklären ist und das hohe Kohlenstoffsenkenpotential dieser Ökosysteme aufzeigt (Pan et al., 2013). Aus diesem Grund wird die Etablierung von Waldflächen auf genutzten Böden, welche aufgrund des degradierten Kohlenstoffvorrates ein besonders hohes Senkenpotential aufweisen, bewusst vollzogen (Lal, 2004; Lorenz and Lal, 2010; Lal et al., 2015; Ussiri and Lal, 2017a).

Der Landbedeckungswandel hin zu Waldvegetation führt zu einer Veränderung des Bodenkohlenstoffvorrates und der qualitativen Zusammensetzung des Bodenkohlenstoffes. Dies

ist vor allem bei der durch den Menschen induzierten Umwandlung von intensiv genutzten Landbedeckungstypen zu forstwirtschaftlich attraktiven Waldsystemen untersucht worden (Guo and Gifford, 2002; Vesterdal et al., 2002; Poeplau et al., 2011; Poeplau and Don, 2013; Bárcena et al., 2014a, 2014b; Guidi et al., 2014a). Die Studien zeigen, dass die vormalige Landbedeckung sowie der Waldtyp mitbestimmend über den Verlauf der Bodenkohlenstoffvorratsveränderung sind und die Zunahme der Waldvegetation eine Erhöhung des partikulären organischen Materials und eine Reduktion des Kohlenstoffs, welcher mit der mineralischen Bodenphase in Verbindung steht, verursacht.

Die quantitativen und qualitativen Veränderungen der Bodenkohlenstoffeigenschaften infolge der Zunahme von Buschvegetation in Grenzräumen ist bis anhin jedoch noch nicht untersucht worden. Aus diesem Grund konzentriert sich die vorliegende Dissertation auf Grenzökotone in alpinen oder subarktischen Gebieten, in denen der Landbedeckungswandel durch das Einwachsen von Buschvegetation stattfindet (Tape et al., 2006; Gehrig-Fasel et al., 2007; Montané et al., 2007; Aradóttir, 2007; Cioldi et al., 2010; Myers-Smith et al., 2011; Caviezel and Kuhn, 2012; Huber and Frehner, 2013). Bei der Auswahl der Untersuchungsgebiete sind gezielt Prozesse für den Landbedeckungswandel ausgewählt worden, die nicht wie in den im Abschnitt zuvor zitierten Studien durch eine bewusste und aktive Landnutzungsänderung gesteuert werden. Dabei ist das Verhalten des Bodenkohlenstoffes (0-30 cm) infolge der Verbuschung in drei Fallstudien untersucht worden. Die Methoden, welche dazu verwendet worden sind, beinhalten die Bestimmung des Bodenkohlenstoffvorrates (Aalde et al., 2006a; Ellert et al., 2008; Rodeghiero et al., 2009) und Beschreibung der Bodenkohlenstoffqualität, welche in dieser Arbeit als Funktion der Kohlenstoffspeicherung betrachtet wird. Bei Vegetationsänderungen hin zu Busch- oder Waldvegetation eignen sich laut Jandl et al. (2014) physikalische Separierungstechniken, um die Qualität des Bodenkohlenstoffs zu beschreiben. Daher ist in der vorliegenden Arbeit der Feinboden inkl. dessen Kohlenstoff nach Grösse und Dichte separiert worden (Zimmermann et al., 2007b). Mit dieser Methode lässt sich der Kohlenstoff in die Fraktionen „POM“ (partikuläre, für Zersetzer leicht zugängliche organische Substanz), „HF“ (in der Sand- und Aggregatfraktion gespeicherter Kohlenstoff), „S+C“ (in der Schluff- und Tonfraktion gespeicherter Kohlenstoff) und „DOC“ (gelöster organischer Kohlenstoff) unterteilen. Damit kann eine eine Abschätzung zur Verweilzeit des Kohlenstoffs im Boden gemacht werden.

Die räumliche und zeitliche Verbuschungsdynamik von subalpinen Alpweiden in den Alpen durch die sich schnell ausbreitende und produktive Grünerle (*Alnus viridis* (Chaix) DC.) ist in Kapitel 2 untersucht worden. Die Zunahme der Buschfläche zwischen 1959 und 2007 beträgt 87 ha (+ 63 %). Die Resultate, welche durch Verrechnung der Luftbildanalyse mit Reliefparametern erzielt worden sind, zeigen in deutlicher Weise auf, dass *A. viridis* im Unteralptal nicht nur an den laut der Literatur potentiellen Standorten (feuchte und nordexponierte Hänge oder Stellen mit hoher Geomorphodynamik z.B. Murgangrinnen) eingewachsen ist. Neben den 73 ha der 400 ha, welche die Fläche der definierten ökologischen

Nische darstellen, wächst die Buschart ausserhalb auf 150 ha. Die zeitliche Analyse bringt hervor, dass die Angaben aus der Literatur für historische Ausbreitungszustände zutreffend sind, sich *A. viridis* jedoch innerhalb der vergangenen 50 Jahre auch auf weniger stark geneigten Hängen (< 60 %), auf strahlungsgünstigeren Südwest- und Südosthängen und auf geomorphodynamisch weniger aktiven Flächen (wie Schutthänge, Schuttkegel) ausgebreitet hat. Die Studie kommt zum Schluss, dass die ökologische Nische von *Alnus viridis* (Chaix) DC. grösser ist, als bisher angenommen und die Landaufgabe nebst dem Relief ein entscheidender Faktor bei der Ausbreitung von *A. viridis* ist. Diese Resultate über das Verhalten der Grünerle im Unteralptal sind daher vergleichbar mit anderen Studien aus dem alpinen Raum, welche das im Vergleich zu anderen Buscharten schnelle Ausbreitungsmuster festgestellt haben. Das Einwachsen der Grünerle und die geoökologischen Auswirkungen sind erst Forschungsgegenstand (Anthelme et al., 2003; Wiedmer and Senn-Irlet, 2006; Caviezel et al., 2010; Huber and Frehner, 2013; Caviezel et al., 2014; Hiltbrunner et al., 2014; Meusburger and Alewell, 2014; Bühlmann et al., 2016; Mueller et al., 2016) und die Datengrundlage für beispielsweise das Bodenkohlenstoffverhalten während der Verbuschung durch die Buschart ist kaum vorhanden (FOEN, 2015). Kapitel 3 widmet sich deshalb der Auswirkung der Verbuschung subalpiner Alpweiden durch A. viridis auf den Bodenkohlenstoff.

Auf Basis der Verbuschungsdynamik ist eine Chronosequenz-Studie zur Untersuchung der Veränderung des Bodenkohlenstoffhaushaltes infolge der Verbuschung durch *A. viridis* durchgeführt worden. Hunziker et al. (2017) (Kapitel 3) zeigt auf, dass das Einwachsen von *A.viridis* den Bodenkohlenstoffvorrat quantitativ und qualitativ signifikant verändert. Während den ersten 40 Jahren der Verbuschung durch *A. viridis* nimmt der Gesamtkohlenstoffvorrat (0-30 cm) der Alpweiden von 100 t C ha^{-1} auf 81 t C ha^{-1} ab, weshalb der Boden in diesem Zeitraum als C Quelle (0.48 t C ha^{-1} a^{-1}) agiert. Jedoch beträgt der Kohlenstoffvorrat (0-30 cm) nach 90 jährigem Grünerlenwachstum und der Bildung des Lebensraumtyps „*Alnenion viridis*" 174 t C ha^{-1}, was einer signifikanten Erhöhung des Kohlenstoffreservoirs um 74 % im Vergleich zu jenem der Alpweide (v.a. Lebensraumtyp *Poion alpinae*) entspricht. Der Boden stellt somit zwischen 40 und 90 Jahren nach dem Landbedeckungswandel eine C-Senke dar (1.86 t C ha^{-1} a^{-1}). Über den Zeitraum von 90 Jahren betrachtet, beträgt die jährliche Kohlenstoffzunahme 0.86 t C ha^{-1}. Der Vergleich der relativen Anteile der C-Konzentrationen der einzelnen Fraktionen im Verhältnis zur Gesamtkohlenstoffkonzentration im Boden deutet durch den Anstieg der POM- und DOC-Anteile und den Abnahmen der HF- und S+C-Anteile auf einen Anstieg der SOC Vulnerabilität durch die Etablierung des *Alnenion viridis* auf subalpinen Alpweiden innerhalb von 90 Jahren hin.

Im subarktischen Raum ist der „mountain birch belt" als weiterer Grenzraum als Untersuchungsgebiet ausgewählt worden. In Island ist die Auswirkung der Aufforstung mit *Betula pubescens* Ehrh. auf das Kohlenstoffverhalten von stark degradierten Böden untersucht worden (Kapitel 4). Und in Südwestgrönland ist die Ausbreitung von *Betula pubescens* Ehrh. im

Zusammenhang mit der Klimaerwärmung als Auslöser herangezogen worden, um das Bodenkohlenstoffverhalten zu charakterisieren (Kapitel 5).

Im Süden Islands weisen demnach stark degradierte Böden einen um 20 t C ha^{-1} tieferen Kohlenstoffvorrat auf als jene mit ungestörter und natürlich gewachsener Birkenbuschvegetation (59 t C ha^{-1}), was ein Potential zur C-Speicherung von vegetationslosen Böden darstellt. Die Etablierung von *B. pubescens* Buschflächen infolge der Aufforstung zeigt eine kontinuierliche Zunahme des Bodenkohlenstoffvorrats (0-30 cm) von 31 t C ha^{-1} auf 46 t C ha^{-1} zwischen 15 und 50 jährigen Birkenbeständen auf und dient mit einer jährlichen Speicherrate von 0.43 t C ha^{-1} als C-Senke. Der angestrebte SOC Vorrat von 59 t C ha^{-1} ist nach 50 Jahren Birkenwachstum noch nicht erreicht. Die Etablierung von Birkenvegetation auf den stark degradierten Böden führt dazu, dass die C Konzentration in der POM-Fraktion am stärksten zunimmt und nach 50 Jahren sogar höher liegt als bei natürlichen, ungestörten Birkenbuschwäldern. Die C Konzentrationen in den mineralischen SOC Fraktionen (HF und S+C) nimmt während des Aufkommens von Birkenvegetation zu, was auf eine Stabilisierung des Bodenkohlenstoffes schliessen lässt. Der Vergleich des SOC zwischen den einzelnen Fraktionen zeigt auf, dass trotz absolutem Anstieg der Konzentrationen in der HF- und S+C-Fraktion eine Stagnation in der HF- resp. Abnahme in der S+C-Fraktion des relativen Anteils des SOC in den mineralischen Fraktionen im Verhältnis zum Anstieg des relativen SOC Anteils in der POM-Fraktion während der Entstehung von Birkenbuschwald stattfindet. Die Aufforstung auf degradierten Böden mit *B. pubescens* Ehrh. führt somit in den ersten 50 Jahren zu labileren Bodenkohlenstoffbedingungen. Die Resultate zeigen darüber hinaus, dass die Entwicklung der Standorte ab 50 Jahren Birkenwachstum wieder zu stabileren Bodenkohlenstoffbedingungen führen kann, weil der angestrebte Gleichgewichtszustand innerhalb von 50 Jahren noch nicht erreicht worden ist. Aufgrund der Bodenerosion in historischer Zeit stellt der angewendete Chronosequenz-Ansatz jedoch nicht das geeignetste Beprobungsschema dar, denn die Kohlenstoffvorräte der degradierten Böden, welche als Ausgangszustand betrachtet worden sind, weisen höhere Werte (39 t C ha^{-1}) auf als die neu etablierten Birkenbestände. Mit Hilfe der physikalischen Fraktionierung ist ersichtlich geworden, dass bei stark degradierten Böden zwei Drittel des Kohlenstoffs in der S+C-Fraktion vorliegen, was bei der Inventarisierung des Bodenkohlenstoffes und der Abschnätzung des Speicherpotentials von degradierten Böden in Zukunft berücksichtigt werden muss.

Aufgrund der Klimaerwärmung von 2.5 °C während den letzten 110 Jahren und einer weiteren Zunahme der Temperatur um 3.3 °C bis 2100 wird in Kombination längerer Vegetationsperiode eine Zunahme der Buschvegetation im Boreal-Tundra Grenzökoton in Südwestgrönland vorhergesagt (Normand et al., 2013). Die vorherrschende Art im Tundrawald dieser Gegend ist *Betula pubescens* Ehrh (Böcher, 1979). Die Studie (Kapitel 5) hat somit die Bodenkohlenstoffeigenschaften von Birkenbuschvegetation und buschloser Tundravegetation auf Einzugsgebietsebene verglichen und dabei abgeschätzt, welche Auswirkungen eine Zunahme der Buschvegetation infolge der Klimaerwärmung auf den Bodenkohlenstoff hat. Die

Resultate zeigen, dass die Kohlenstoffvorräte (0-30 cm) von Birkenbuschvegetation und buschloser Vegetation zwischen 54 und 148 t C ha^{-1} variieren. Die Unterschiede sind mehr durch die untersuchten Vegetationsstandorte in der Landschaft zu erklären als durch den Vegetationstyp. Die landschaftstypischen Eigenschaften beeinflussen den Biomassevorrat in der Vegetation und das Angebot an Kohlenstoff für den SOC Vorrat. Der Bodenkohlenstoff wird vorwiegend in der POM- und S+C-Fraktionen gespeichert (absolut und relativ), wobei die POM-Fraktion bei Birkenstandorten und die S+C-Fraktion bei buschlosen Vegetationsstandorten dominierend sind. Wie die Studie hervorbringt, kann die POM-Fraktion aber auch bei buschloser Vegetation einen ähnlichen Anteil an Kohlenstoff wie die S+C-Fraktion speichern, weil die Standorteigenschaften eine Zersetzung der organischen Substanz hindern können. Eine Ausbreitung der Birkenvegetation aufgrund der Klimaerwärmung kann an Standorten, welche für das Wachstum günstig sind, zu einer Erhöhung des Bodenkohlenstoffvorrates führen, was aber mit einer Zunahme der Labilität des SOC verbunden ist.

Unabhängig von den Prozessen, die zu einem Aufkommen der Buschvegetation in marginalen Grenzräumen führen, zeigen die Resultate der drei Fallstudien, dass der Landbedeckungswandel die Quantität und Qualität des Kohlenstoffs im mineralischen Boden verändert. Bestehende „carbon response functions“ für die Umwandlung in Waldsysteme können für den Landbedeckungswandel zu Buschvegetation in alpinen und subarktischen Räumen nicht angewendet werden, weil die Produktivität der Buscharten und vermutlich der Geoökofaktor Temperatur das Bodenkohlenstoffverhalten entscheidend beeinflussen. Alle drei Fallstudien haben eine Zunahme der Konzentration und des relativen Anteils des labilen Kohlenstoffs in der POM- und DOC-Fraktion gemessen. Weiter zeigen die relativen Veränderungen der Konzentrationen in den SOC Fraktionen eine Stagnation oder Abnahme des SOC in der HF-Fraktion. Die Arbeit hat nicht abschliessend beantworten können, ob mit den teils verzeichneten Zunahmen der C-Konzentration in der „Schluff- und Ton“-Fraktion eine Stabilisierung dieses Kohlenstoffs in dieser Fraktion einhergeht.

INHALTSVERZEICHNIS

ABBILDUNGSVERZEICHNIS

TABELLENVERZEICHNIS

KAPITEL 1

Einleitung

Blick aus dem Untersuchungsbiet nördlich von Igaliku (Grönland) Richtung Osten zum Berg Illerfissalik (1752 m ü. M.). Im Vordergrund der Bach, der durch die mit Tundravegetation bewachsene Grundmoränenlandschaft fliesst und das Tal zum Einarsfjord hin entwässert. Aufgenommen von M. Hunziker am 5. Juli 2013.

1.1 Der Landbedeckungswandel infolge der Landnutzungswandels

Menschen leben und bewegen sich in Landschaften, die sie seit dem Paläolithikum genutzt und mitgestaltet haben (Ramankutty et al., 2006). Diese Nutzung lässt sich in drei Phasen gliedern, wobei der Beginn einer neuen Phase stets mit einer Intensivierung des anthropogenen Einflusses auf die Landschaft und deren einzelnen Ressourcenelemente gekoppelt ist (De Vries and Goudsblom, 2002; Turner II and McCandless, 2004).

Die dritte Phase, welche auch als Anthropozän bezeichnet wird (Crutzen, 2002; Steffen et al., 2007), hat vor ungefähr 300 Jahren begonnen und ist durch die Nutzung fossiler Brennstoffe, einem exponentiellen Bevölkerungswachstum, dem Aufkommen des Kapitalismus, einer steigenden Globalisierung und dem Einsatz neuer Technologien infolge der Industriellen Revolution geprägt (Ramankutty et al., 2006). Seit 1945 hat sich die Weltbevölkerung verdreifacht und die Weltwirtschaft ist um das 15-fache gewachsen (Steffen et al., 2007; UN, 2015). Die ökonomische Globalisierung hat seit Anfang der 1980-er Jahren durch die Beseitigung von verschiedensten wirtschaftlichen Barrieren zu einer verstärkten Liberalisierung und Vernetzung der Märkte auf globaler Ebene geführt. Dies wirkt sich seither noch stärker auf die Erschliessung verschiedener Rohstoffe, die Produktion und Veredelung von Gütern, den Verkauf der Endprodukte und das dazu notwendige Transportnetz aus (Lambin et al., 2001; Osterhammel and Petersson, 2012).

Die anthropogene Nachfrage nach Nahrungs- und Futtermitteln, Holz sowie mineralischen Roh- und Brennstoffen hat die Umnutzung der natürlichen Ökosysteme erfordert (Swinton et al., 2007; Lal, 2013). Der globale Landnutzungswandel hat zwischen 1700 und 1990 zu einer Abnahme der Landbedeckungstypen „Wald“ um 13 Mkm^2 (- 24 %), „Steppe/Savanne/Grasland“ um 15 Mkm^2 (- 45 %), „Buschland“ um 6 Mkm^2 (- 71 %) und „Tundra/Wüsten“ um 4 Mkm^2 (- 14 %) Mkm^2 geführt. Im selben Zeitraum haben die Flächen von „Ackerbauland“ um 12 Mkm^2 (+ 544 %) und „Weideland“ um 26 Mkm^2 (496 %) Mkm^2 zugenommen (Goldewijk, 2001). Dabei sind die Ökosysteme der temperierten Laubwälder in Europa und im Osten der USA sowie jene der tropischen Laubwäldern in Südasien grösstenteils in Ackerland umgewandelt worden (Ramankutty and Foley, 1999; Ramankutty et al., 2008). Natürliche Steppen- / Savannen- und Graslandflächen sind dagegen als Weideflächen umgenutzt worden (Ramankutty and Foley, 1999; Ramankutty et al., 2008). Regionale Unterschiede hinsichtlich des Landnutzungswandels sind in Ramankutty et al. (2008) aber auch identifiziert worden. Nach Foley et al. (2005) stehen zwei Drittel der terrestrischen Oberfläche unter landwirtschaftlicher oder anderer anthropogener Nutzung. Die gesellschaftlichen, wirtschaftlichen und wirtschaftspolitischen Entwicklungen zeigen deutlich auf, wie der „Mensch“ als Regler (Leser and Mosimann, 1997) während des Anthropozäns im natürlichen Geosphärensystem auf globaler Ebene Einfluss genommen hat und sich das Geosphärensystem in entscheidender Weise wandelt (Millennium Ecosystem

Assessment, 2005; Costanza et al., 2007). Ein essentieller Aspekt davon ist die Veränderung des aktiven Kohlenstoffkreislaufes, welche Rückkopplungseffekte auf andere Prozesse hat.

1.2 Auswirkungen auf den aktiven Kohlenstoffkreislauf

Seit dem Beginn der Industrialisierung wird dem aktiven Kohlenstoffkreislauf, dessen Stoffflüsse zwischen der Atmosphäre, dem terrestrischen System (bestehend aus Biomasse und Boden) und den Ozeanen stattfinden, fossiler Kohlenstoff aus dem passiven resp. langfristigen Kohlenstoffkreislauf zugeführt. Dies führt zu einem Nettoanstieg der Kohlenstoffmenge im aktiven resp. kurzfristigen Kohlenstoffkreislauf (Ussiri and Lal, 2017b). Dieser fossile Kohlenstoff, der als Energiequelle genutzt wird, hat sich zwischen 360 und 65 Ma vor heute während dem späten Paläozoikum (Karbon und Perm) sowie in weniger starker Ausprägung im Mesozoikum gebildet und ist dem kurzfristigen Kreislauf in der damaligen Zeit entzogen worden (Houghton, 2014; Montañez, 2016). Auf einer viel kleineren Zeitskala von Jahrzehnten bis Jahrhunderten wird dieser fossile Kohlenstoff seit dem 18. Jahrhundert wieder dem aktiven Kreislauf zugeführt. Zusätzlich zur Nutzung des fossilen Kohlenstoffs haben innerhalb des aktiven Kohlenstoffkreislaufes die landwirtschaftliche Urbarmachung und die Landnutzungsänderungen (Kaptiel 1.1) zur Umverteilung des Kohlenstoffs zwischen den oben erwähnt Reservoiren geführt (Ciais et al., 2013).

Aktuelle Hochrechnungen bilanzieren den weltweiten Ausstoss von Kohlenstoff in die Atmosphäre für den Zeitraum zwischen 1750 und 2015 auf 600 ± 70 GtC. Davon stammen 410 ± 20 GtC von den fossilen Brennstoffen sowie der Zementindustrie und 190 ± 65 GtC von der Landnutzungsänderung (Le Quéré et al., 2016) (Abbildung 1). Für die terrestrischen Systeme bedeutet die Abnahme des Kohlenstoffvorrates, dass sich durch die Umnutzung das Kohlenstoffgleichgewicht auf einem tieferen Niveau einstellt. Beispielsweise ist der Gleichgewichtszustand bei der Landnutzungsänderung von Waldvegetation zu Ackerland nach 23 Jahren auf einem um 31 % tieferen Niveau erreicht worden (Poeplau et al., 2011). Durch diese zusätzlichen Kohlenstoffflüsse haben die Vorräte in der Atmosphäre um 260 ± 5, im Ozean um 175 ± 20 und im terrestrischen System um 165 ± 70 GtC zugenommen (Le Quéré et al., 2016) (Abbildung 1).

1.3 Auswirkungen des veränderten Kohlenstoffkreislaufs

Laut Kyoto Protokoll werden die gasförmigen Kohlenstoffmolekülverbindungen Kohlendioxid (CO_2) und Methan (CH_4) als Treibhausgase definiert (Field et al., 2013). Im Anthropozän haben die oben beschriebenen Veränderungen der Kohlenstoffflüsse dazu geführt, dass die Kohlenstoffdioxidkonzentration (CO_2) in der Atmosphäre zwischen 1750 und 2015 von 270 auf 400 ppm zugenommen hat. Die Konzentrationen von Methan (CH_4) ist während desselben

Zeitraums von 700 auf 1845 ppb angestiegen (Ciais et al., 2013; Le Quéré et al., 2016; Ussiri and Lal, 2017c). Die deutliche Zunahme dieser Treibhausgasekonzentrationen, welche seit den 1950er Jahren gemessen werden, hat zum anthropogenen Treibhauseffekt geführt (Le Quéré et al., 2016).

Die Folgen der veränderten Kohlenstoffflüsse sind messbar. Die globale Durchschnittstemperatur der Oberfläche (über Land und Ozean) ist zwischen 1880 und 2015 um 0.96 °C angestiegen (Dahlman, 2017).

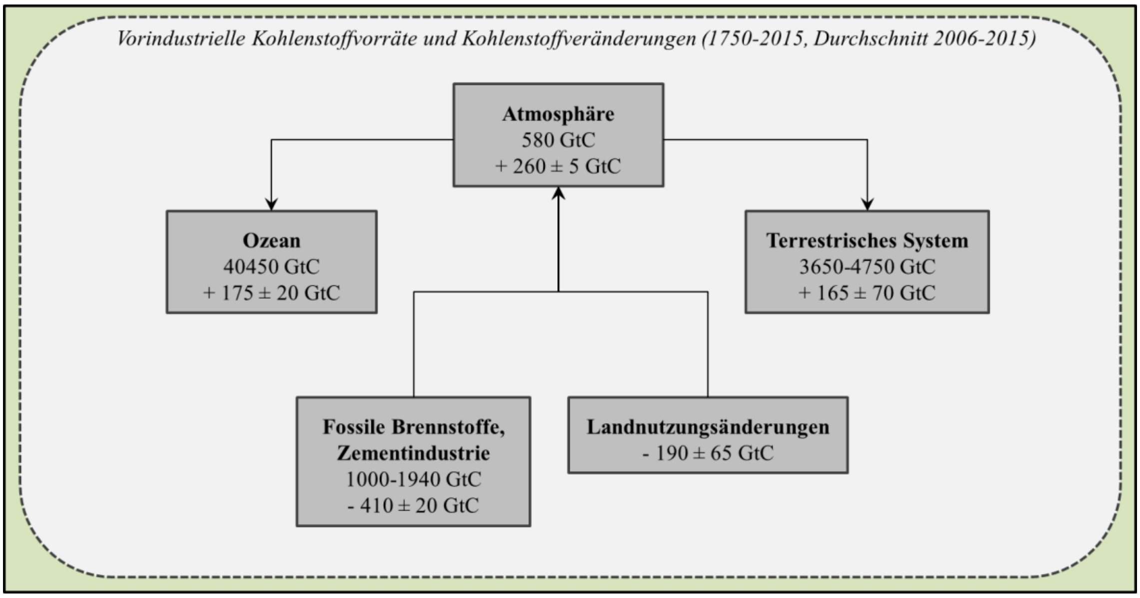

Abbildung 1: Der aktive Kohlenstoffkreislauf zwischen Atmosphäre, Ozean, Lithosphäre und terrestrischem System. Aufgeführt werden die abgeschätzten, vorindustriellen (um 1750) Kohlenstoffvorräte (Ciais et al., 2013) und darunter die aufsummierten Kohlenstoffveränderungen in den einzelnen Kohlenstoffreservoirs zwischen 1750 und 2015 (Le Quéré et al., 2016). Die Kohlenstoffflüsse zwischen den einzelnen Reservoirs resultieren aus Messungen und Modellierungen (Le Quéré et al., 2016). 1 GtC entspricht 1 PgC oder 3.664 Gt CO_2.

Nebst der Klimaerwärmung, welche bis ins Jahr 2100 um weitere 1.4 bis 5.8 °C zunehmen kann (IPCC, 2014a), wirken sich diese Änderungen auf das globale Geosphärensystem, aber auch auf regionale Geosysteme aus. Dies ist beispielsweise durch folgende Parameter erfassbar: Meeresspiegelhöhe, Salinität der Ozeane, Wassertemperatur, Ozonkonzentration in der Atmosphäre, Strahlungshaushalt, atmosphärischer Wasserdampf, Luftdruck, Niederschlagsverhalten, Windgeschwindigkeiten, terrestrische Albedo, Permafrost, terrestrischer Wasserkreislauf (Field et al., 2013; Blunden and Arndt, 2015). Diese Veränderungen wirken sich auch auf die floristische und faunistische Biodiversität und die Bodenqualität in Richtung Bodendegradation aus, welche wiederum die Ökosystemdienstleistungen und die Nahrungsmittel- sowie Trinkwassersicherheit beeinflusst (Lambin et al., 2001).

Aus diesen Gründen wird versucht, den Netto-Kohlenstofffluss und die Flüsse anderer Treibhausgase in die Atmosphäre zu minimieren. Betreffend des Kohlenstoffs wird zudem angestrebt, den Vorrat im Reservoir „Atmosphäre" zu reduzieren, indem die Stoffflüsse von der

Atmosphäre in die anderen Reservoire erhöht werden (IPCC, 2014b). Laut Le Quéré et al. (2009) haben die Ozeane und das terrestrische System zwischen 1959 und 2008 jährlich zwischen 40 und 45 % des jährlich in die Atmosphäre ausgestossenen CO_2 aufgenommen. Daher ist das Senkenpotential des terrestrischen Systems von hohem Interesse. Es kann als noch effektivere Senke fungieren, wenn sein Potential, atmosphärisches CO_2 mittels Photosynthese in der Biomasse der Vegetation zu speichern, erhöht wird (Lal, 2008). Diese Massahme ermöglicht es, in kurzer Zeit (Jahrzehnten bis Jahrhunderten) auf natürliche Weise atmosphärischer Kohlenstoff in der Biomasse und später über Einarbeitungsprozesse im Boden kostengünstig zu speichern (Ussiri and Lal, 2017c).

Der Boden als Kohlenstoffspeicherreservoir weist wegen der Speicherfähigkeit und der Menge an enthaltenem Kohlenstoff ein grosses Potential auf, zusätzlichen organischen Kohlenstoff zu speichern. Denn der Boden als Teil des terrestrischen Systems enthält mehr als doppelt soviel Kohlenstoff (1500 bis 2400 GtC) als in der Atmosphäre (590 GtC) vorhanden ist (Batjes, 1996; Ciais et al., 2013). Hinsichtlich der CO_2 Reduktion in der Atmosphäre haben deshalb kleinste Zunahmen des Bodenkohlenstoffes grosse Auswirkungen auf den Kohlenstoffvorrat in der Atmosphäre (Lal, 2004; Smith, 2004).

Dabei ist der Landnutzungswandel von Vegetationstypen mit vergleichsweise tiefem Kohlenstoffvorrat hin zu Typen mit höherem Vorrat eine Massnahme mit grossem Speicherungspotential (IPCC, 2014b; Ussiri and Lal, 2017c). Die Nettoprimärproduktion eines Systems wird dadurch erhöht, indem beispielsweise ein sekundäres Waldsystem durch Aufforstung auf Grasland oder wüstenhafter Landschaft entsteht (Ussiri and Lal, 2017a). Im Vergleich zu den Landbedeckungsänderungen, welche in Kapitel 1.1 aufgeführt worden sind, handelt es sich hierbei um den gegenteiligen Prozess, bei welchem es zu einer Speicherung von Kohlenstoff in der Biomasse und im Boden kommt.

1.4 Die Bestimmung und Überwachung des terrestrischen Kohlenstoffsenkenpotentials

Auf globaler Ebene wird jährlich das Budget der Kohlenstoffflüsse berechnet, indem die Kohlenstoffemissionen durch Industrie (E_{FF}) und des Landnutzungswandels (E_{LUC}) mit den Aufnahmeraten der Senken Ozean (S_{OCEAN}), terrestrisches System (S_{LAND}) und Atmosphäre (G_{ATM}) bilanziert werden ($G_{ATM} = E_{FF} + E_{LUC} – S_{OCEAN} - S_{LAND}$) (Le Quéré et al., 2016). Entlang der Zeitachse hat der jährliche Ausstoss von fossilen Brennstoffen und der Industrie seit Mitte des 20. Jahrhunderts stark zugenommen und beträgt heute 10 GtC a^{-1} (Abbildung 2). Dabei verhält sich der Kohlenstofffluss E_{LUC} stagnierend bei 1-2 GtC a^{-1} und ist in den vergangenen Jahren sogar etwas rückläufig. Bei den Senken nimmt der Fluss von S_{OCEAN} leicht zu (2 GtC a^{-1}), jener von G_{ATM} weist eine äusserst variable Zunahme auf (zuletzt 6 GtC a^{-1}). Die Erhöhung des Kohlenstoffreservoirs im terrestrischen System von 3 GtC pro Jahr, die trotz dem Verlust an

Kohlenstoff durch die Landnutzungsänderungen (Kapitel 1.1), resultiert, ist der Überschuss, der bei der Berechnung des globalen Kohlenstoffbudgets entsteht und als C-Senke im terrestrischen System betrachtet wird (Raupach and Canadell, 2010; Houghton, 2014; Le Quéré et al., 2016) (Abbildung 2).

Die Quantifizierung der Kohlenstoffveränderungen im terrestrischen System (Vegetation und Boden) ist schwierig. Aufgrund von regionalen Unterschieden in den Prozessen der Kohlenstoffspeicherung werden unterschiedliche Erfassungsmethoden und Modelle verlangt. Zusätzlich haben historische Gründe zu einer Vielzahl von verschiedenen Methoden geführt. Die Erfassung der Biomasseveränderungen und damit verbunden die Freisetzung oder Speicherung von Kohlenstoff wird gegenwärtig mittels top-down Ansatz mit Satellitenaufnahmesystemen und Klima-Vegetationsmodellen, sogenannten „dynamic global vegetation models“ (DGVM), oder Biomasseinventaren (bottom-up Ansatz) erfasst (Ramankutty et al., 2006; Houghton, 2014).

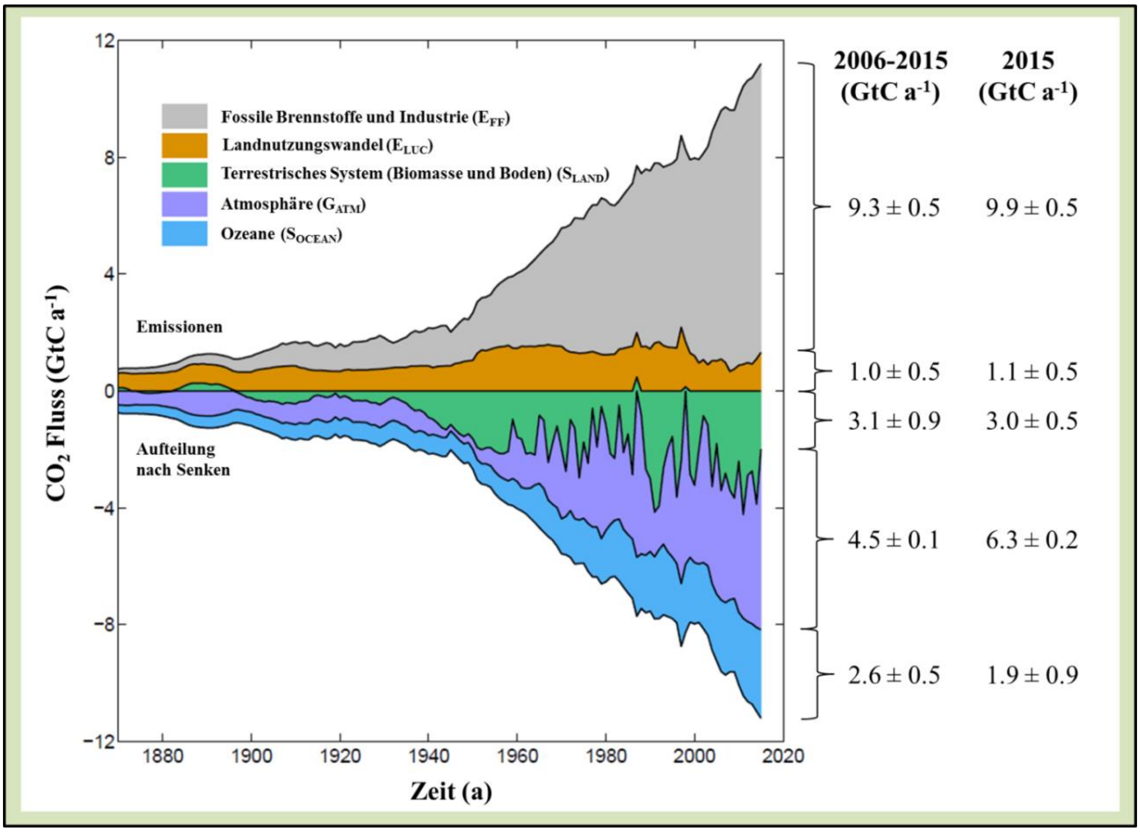

Abbildung 2: Das zeitlich aufgelöste der Veränderung des globalen Kohlenstoffbudgets zwischen 1880 und 2015. Das Jahr 1959 stellt den Übergang zwischen der Abschätzung der Veränderungen auf Basis von historischen Daten (vor 1959) und der Modellierungen der jährlichen Veränderungen auf Basis von Messdaten (nach 1959) dar (Le Quéré et al., 2016, verändert).

Die Biomasse des terrestrischen Kohlenstoffvorrats und dessen Veränderung kann mit guter zeitlicher Auflösung abgeschätzt werden. Im Vergleich dazu sind die Bilanzierungen des Vorrats und dessen Wandel in den Böden mit grösserem Aufwand verbunden und auf aktuellem Stand zu halten. Dafür gibt es Gründe. Die Generierung eines statistisch fundierten Datensatzes ist in der Erhebung kosten- und zeitintensiv (Stockmann et al., 2013), weil idealerweise SOC Vorratsveränderungen durch den Landbedeckungswandel mit einer Neubeprobung derselben Böden verbunden sind (Aalde et al., 2006a). Da dies nicht immer möglich ist, beruhen Inventare und Modelle (unabhängig der räumlichen Skala) vielmals auf Annahmen sowie älteren, bereits bestehenden SOC Datensätzen (z.B. Nussbaum et al., 2014; Stockmann et al., 2015). Deshalb konzentrieren sich die Bemühungen um die Quantifizierung der SOC Vorräte auf besiedelte Regionen und anthropogen genutzten Landbedeckungstypen wie „Ackerland“ und „Weideland“. Natürliche Landbedeckungstypen oder Typen von Landbedeckungswandel, welche weniger stark durch das menschliche Handeln induziert sind, werden weniger stark priorisiert. Als Folge davon bestehen global angelegte Datensätze (z.B. WISE Datenbank v.3.1) zwar aus zahlreichen Bohrprofilen (10250 Einträge), wobei nicht ackerbaulich genutzte Gebiete Nordamerikas, die nordischen Länder, die meisten Länder Asiens, Nordafrika und Australien verhältnismässig untervertreten sind (Köchy et al., 2015; Hengl et al., 2017). Über die Aussagekraft und die Genauigkeit solcher Karten hinsichtlich der Wiedergabe der SOC Vorräte in der Landschaft ist im Moment zu spekulieren. Um die Genauigkeit der Bestimmung des Kohlenstoffeigenschaften der Böden zu verbessern, besteht ein grosser Bedarf, die Datengrundlage in Form von harten Messdaten zu verbessern, insbesondere bei der Untersuchung des SOC infolge der Landbedeckungsveränderungen in Landschaften mit geringem Nutzungsdruck (Canadell et al., 2010; Jandl et al., 2014).

1.5 Kohlenstoff in Waldsystemen und dessen Inventarisierung

Wälder bedecken 30 % der terrestrischen Erdoberfläche und stellen im Vergleich zu den anderen Landbedeckungstypen (Kap. 1.1) den flächenmässig grössten Vegetationstyp dar (FAO, 2010). Weil global betrachtet 75 % der Bruttoprimärproduktion den Waldsystemen zuzuordnen sind (Beer et al., 2010), weisen sie zusammen eine jährliche Kohlenstoffspeicherungsrate von 2.4 GtC a^{-1} (80% der globalen Kohlenstoffspeicherung, siehe Kapitel 1.4) auf (berechnet über den Zeitraum 1990-2007) (Pan et al., 2011). Sie agieren daher als wichtige Steuergrösse bei der Reduktion von atmosphärischem CO_2 (Abbildung 2). Diese Systeme enthalten 70 bis 90 % der weltweit vorhandenen ober- und unterirdischen Biomasse, welche somit im terrestrischen System mit 360 GtC ein wichtiges Kohlenstoffreservoir bildet (Kindermann et al., 2008; Ciais et al., 2013; Lorenz, 2013; Ussiri and Lal, 2017a). Zudem beinhalten laut Ussiri und Lal (2017a) die Waldsysteme weltweit den grössten Bodenkohlenstoffvorrat (500 GtC). Zusammen mit der Vegetation bilden sie den grössten Kohlenstoffspeicher im terrestrischen System, welcher sogar den Kohlenstoffvorrat in der Atmosphäre übersteigt.

Laut FAO (2006a) werden Flächen, die mit Baumvegetation bestockt sind als Wald definiert, wenn die Fläche > 0.5 ha gross ist, die Bäume > 5 m hoch sind und das Kronendach einen

Bedeckungsgrad von > 10 % besitzt. Dies bedeutet jedoch auch, dass Flächen mit holziger Übergangsvegetation wie Buschvegetation in Grenzräumen wie im alpinen oder subarktischen Raum nicht als „Wald“ klassifiziert werden. Solche Vegetationstypen erhalten deshalb bei Inventarisierungprogrammen von Biomasse- und Bodenkohlenstoff für ihre direkte Beprobung eine geringere Priorität als die Waldflächen. In vielen Fällen werden ihnen Referenzwerte oder Werte, welche den Kohlenstoffvorrat von Waldtypen repräsentieren, zugewiesen (Nielsen et al., 2011; FOEN, 2014). Auch werden sie bei den Untersuchungen der Bodenkohlenstoffveränderungen infolge der Waldausbreitung nicht berücksichtigt (Poeplau et al., 2011; Poeplau and Don, 2013; Bárcena et al., 2014a). Selbst in montanen Räumen liegt der Fokus der Inventarisierung der Kohlenstoffvorräte bei den forstwirtschaftlich nutzbaren Nadelholzarten, welche durch Aufforstung oder Waldzunahme nach der Landaufgabe auf Grasflächen aufkommen (Risch et al., 2008; Hiltbrunner et al., 2013; Guidi et al., 2014a).

Der tiefe Stellenwert der Übergangsvegetationen in den Grenzräumen ist vermutlich mitunter ein Grund, weshalb der systematische Fehler bei nationalen oder kontinentalen Kohlenstoffbudgets bei 50 % liegt (Canadell et al., 2010). Aus diesem Grund wird eine Anpassung der Methodenwahl gefordert (Canadell et al., 2010; Jandl et al., 2014). Auf globaler Ebene und mittels top-down Ansatz können durch neue Satellitentechniken (MODIS-based observation) nebst den bisherigen vier Waldkategorien, welche die physiognomischen Baumvegetation abbilden, sieben weitere Biome voneinander unterschieden und identifiziert werden, welche aus Baum- und Buscharten bestehen (Pan et al., 2013). Weiter werden DGV-Modelle zur zeitlichen Vegetationsanalyse und damit verbunden zur Simulation der globalen Kohlenstoffflüsse verwendet (Le Quéré et al., 2016). Die Berücksichtigung der marginalen Räume mit der Baum- und Buschvegetation bei der Inventarisierung der Kohlenstoffvorräte als bottom-up Ansatz dient dazu, den systematischen Fehler weiter zu verringern und so das Kohlenstoffbudget zu verbessern.

1.6 Beschreibung der Bodenkohlenstoffveränderung während der Etablierung von Waldsystemen

Der gesamte organische Bodenkohlenstoffvorrat innerhalb einer definierten Bodenschicht (z.B. 0-30 cm) ist durch den Eintrag und Austrag von organischen Substanzen gesteuert (Guo and Gifford, 2002; Vesterdal et al., 2013). Befindet sich der SOC-Vorrat im Gleichgewichtszustand, sind der ober- und unterirdische Eintrag an organischer Substanz sowie der Austrag über die Bodenatmung infolge der Zersetzung organischer Substanz durch das Edaphon und über die Bodenlösung konstant (Six et al., 2002b). Der Landbedeckungswandel von intensiv genutzten Landnutzungstypen hin zu Wald- oder Buschvegetation ändert den Kohlenstoffvorrat (Post and Kwon, 2000; Vesterdal et al., 2002; Poeplau et al., 2011; Vesterdal et al., 2011, 2013; Bárcena et al., 2014a). Dies bedeutet, dass der Gleichgewichtszustand des

SOC-Vorrats gestört wird, bis sich ein neues Gleichgewicht eingestellt hat (Six et al., 2002b). Beim Wechsel von Grasland zu Waldvegetation dauert dies mehr als 150 Jahre und das Gleichgewicht stellt sich auf tieferem Niveau ein (Poeplau et al., 2011). Weitere Meta-Analysen (Guo and Gifford, 2002; Laganière et al., 2010; Bárcena et al., 2014b) zeigen, dass die vormalige Landbedeckung (in diesem Fall Ackerland, Grasland, Heideland oder Boden ohne Vegetation) sowie der Waldtyp (Nadel-, Laub- oder Mischwald) mitbestimmend über den Verlauf der Bodenkohlenstoffvorratsveränderung (C-Senke oder C-Quelle) sind. Einzelstudien aus den alpinen und subarktischen Räumen haben eine Abnahme (Guidi et al., 2014b), eine Abnahme gefolgt von einer Zunahme (Cerli et al., 2006; Hiltbrunner et al., 2013) oder eine Stagnation (Vesterdal et al., 2007) des mineralischen SOC-Vorrates infolge der Waldzunahme gemessen, wobei unterschiedliche Betrachtungszeiträume gewählt worden sind.

Das Aufkommen und Einwachsen von Baum- und Buscharten bewirkt, dass nebst der Änderung der oben genannten Steuergrössen (Hunziker et al., 2014; Guidi et al., 2015; Bühlmann et al., 2016) auch die Anpassung weiterer physikalischer und chemischer Bodenparameter stattfindet (Myers-Smith and Hik, 2013; Caviezel et al., 2014). Dies verändert die Bodenkohlenstoffqualität (Poeplau and Don, 2013). Bei einem Landbedeckungswandel ist somit nicht nur die Untersuchung des Kohlenstoffvorrates sondern auch die Betrachtung der Kohlenstoffqualität notwendig (Jandl et al., 2014).

Bei der Stabilisierung der organischen Substanz im Boden stellen die intristische Rekalzitranz in den ersten Phasen der Zersetzung sowie die räumliche Trennung und die Bildung von organo-mineralischen Komplexen in der späteren Phase der Dekomposition wichtige Mechanismen dar, um die organische Substanz vor der Mineralisierung zu schützen und somit längerfristig im Boden speichern zu können (Schmidt et al., 2011; von Lützow et al., 2006). Zur Untersuchung der unterschiedlich stabilisierten Bodenkohlenstoffgruppen eignen sich im Zusammenhang mit Landbedeckungsänderungen Methoden der physikalischen Bodenfraktionierung (Elliott and Cambardella, 1991; Cambardella and Elliott, 1992; Six et al., 1998; Christensen, 2001). Dazu wird die Separierung nach Grösse und / oder Dichte verwendet, um den Gesamtkohlenstoff verschiedenen Fraktionen zuweisen zu können, in welchen sich die organische Substanz hinsichtlich ihrer Stabilität unterscheiden lässt (Six et al., 2002a; von Lützow et al., 2007; Zimmermann et al., 2007c; Kögel-Knabner et al., 2008). Studien, welche die erwähnten Methoden angewendet haben, haben gezeigt, dass es während der Waldzunahme zu einer Vergrösserung des Menge an partikulärer organischer Substanz (POM) und einer Verringerung der Menge an Kohlenstoff in der organo-mineralischen Fraktionen kommt (Poeplau and Don, 2013; Guidi et al., 2014a).

Die Auswirkungen des Landbedeckungswandels hin zu Waldvegetation zeigen keinen klaren Trend hinsichtlich des Senken- oder Quellenpotentials (Guo and Gifford, 2002; Bárcena et al., 2014b; Poeplau et al., 2011; Vesterdal et al., 2013). Ausserdem ist das Einwachsen von

Buschvegetation in alpinen und subarktischen Grenzräumen und deren Auswirkungen auf den Bodenkohlenstoffvorrat sowie die Qualität des Bodenkohlenstoffes noch nicht erforscht worden.

1.6 Motivation und Ziele

Die Einleitung in das Thema hat die Wichtigkeit der Böden als Kohlenstoffreservoir im Kontext zur globalen Kohlenstoffproblematik aufgezeigt. Das grösste Potential der Kohlenstoffspeicherung im terrestrischen Ökosystem stellt dabei der Wandel hin zu Busch- und Waldvegetation dar (Lorenz, 2013). Wälder enthalten ungefähr 70 bis 90% der Biomasse des terrestrischen Systems und die Speicherungsrate von atmosphärischem Kohlenstoff hat in den vergangenen beiden Jahrzehnten ungefährt 2.5 GtC a^{-1} betragen (Pan et al., 2011; Lorenz, 2013). Der von der Vegetation in den Boden gelangte organische Kohlenstoff hat das Potential, während Tausenden von Jahren gespeichert zu werden (Lorenz, 2013). Momentan ist unklar, wie sich der Bodenkohlenstoff besonders während der Landbedeckungsänderung hin zu Buschvegetation in Grenzräumen hinsichtlich der Quantität und Qualität verhält. Denn vergangene Untersuchungen haben sich auf Bodenkohlenstoffveränderungen zwischen aktiv genutzten Vegetationssystemen und deren Veränderungen konzentriert (Poeplau et al., 2011; Poeplau and Don, 2013; Bárcena et al., 2014b; Köchy et al., 2015). Doch auch in Grenzräumen, die hinsichtlich der aktiven Nutzung als wenig intensiv oder nicht genutzt charakterisiert werden können, sind aufgrund von direkten und indirekten Rückkopplungsmechansimen zwischen den Regelkreisen in dynamischen Systemen (Leser, 2010), während den letzten Jahrzehnten Landbedeckungsänderungen festgestellt worden (Gehrig-Fasel et al., 2007; Montané et al., 2007; Myers-Smith et al., 2011; Caviezel et al., revised).

Die vorliegende Dissertation widmet sich deshalb dem Bodenkohlenstoffverhalten in marginalen Grenzräumen, in denen Landbedeckungsveränderungen festgestellt werden, was Auswirkungen auf den Bodenkohlenstoffhaushalt haben kann. Folgende zwei Grenzräume sind dabei ausgewählt worden:

- die subalpine Vegetationsstufe der Alpen, welche als ökologische Höhenstufe zwischen dem geschlossenen Hochwald und dem alpinen Rasen definiert ist (Leser, 2010) und
- der „mountain birch belt“ im nordschen Raum (Wielgolaski, 2001) als Teil der zirkumpolaren Buschtundra, die als Übergangszone zwischen den beiden Biomen Boreal und Tundra gilt (Payette et al., 2001).

Bei den Gründen, die den Landbedeckungswandel in diesen Grenzräumen auslösen, sind absichtlich Prozesse gewählt worden, die nicht durch eine bewusste und aktive Landnutzungsänderung gesteuert werden, aber eine indirekte Folge der anthropogenen Eingriffe in das Geosphärensystem darstellen. Die berücksichtigen Prozesse und die damit verbundenen Landbedeckungsänderungen sind in drei Fallstudien bearbeitet worden und lauten (Abbildung 3):

- In der Schweiz führen die Extensivierung der Landwirtschaft und die Landaufgabe zur Verbuschung der Alpweiden durch die Grünerle (*Alnus viridis* (Chaix) DC.) (Fallstudie „Landaufgabe").
- In Island werden Flächen mit stark degradierten Vulkanböden, die sich in den Bodeneigenschaften und daher im Kohlenstoffspeicherverhalten zu anderen mineralischen Böden unterscheiden, mit der einheimischen Moorbirke (*Betula pubescens* Ehrh.) aufgeforstet (Fallstudie „Aufforstung").
- Im Südwesten Grönlands bewirkt die messbare Klimaerwärmung eine Ausbreitung der Buschvegetation durch die Moorbirke (*Betula pubescens* Ehrh.) im Boreal-Tundra Grenzökoton (Fallstudie „Klimaerwärmung").

Die übergeordnete Leifrage der Dissertation lautet (Abbildung 3):

Wie verändert sich der organische Kohlenstoff im mineralischen Boden infolge des Landbedeckungswandels?

Die Arbeit hat dabei den Bodenkohlenstoff in quantitativer und qualitativer Hinscht untersucht. Aufgrund der naturräumlichen Standorteigenschaften inkl. den Gründen für den Landbedeckungswandel werden in den Fallstudien (Kapitel 3-5) weitere jeweiligen dem Landschaftssystem angepasste Forschungsfragen definiert und in den Studien beantwortet.

Fallstudie „Landaufgabe"

- Wie verhält sich der Bodenkohlenstoffvorrat während der Verbuschung durch die Grünerle?
- Ist der Bodenkohlenstoffvorrat von Grünerlenbeständen vergleichbar mit dem im nationalen Treibhausgasinventar für Gebüschwald angenommenen Bodenkohlenstoffvorrat von 69 t C ha^{-1}?
- Wie verändert sich die Bodenkohlenstoffqualität während der Verbuschung?
- Wie können sich die Beziehungen zwischen den Eigenschaften des Bodenkohlenstoffes und anderen Bodeneigenschaften verändern und welche Auswirkungen hat dies auf die Prozesse im Landschaftshaushaltssystem?

Fallstudie „Aufforstung"

- Welche Auswirkung hat in Island die Aufforstung mit *B. pubescens* Ehrh. auf das langzeitige Kohlenstoffspeicherpotentials von stark degradierten Vulkanböden?

Fallstudie „Klimaerwärmung“

- Wie unterscheiden sich der Bodenkohlenstoffvorrat und die Qualität des Bodenkohlenstoffes zwischen Buschvegetation mit *B. pubescens* Ehrh. und buschloser Tundravegetation auf Einzugsgebietsebene?
- Welche Parameter kontrollieren die Ausbreitung der Buschvegetation und das Buschwachstum auf Einzugsgebietsebene?

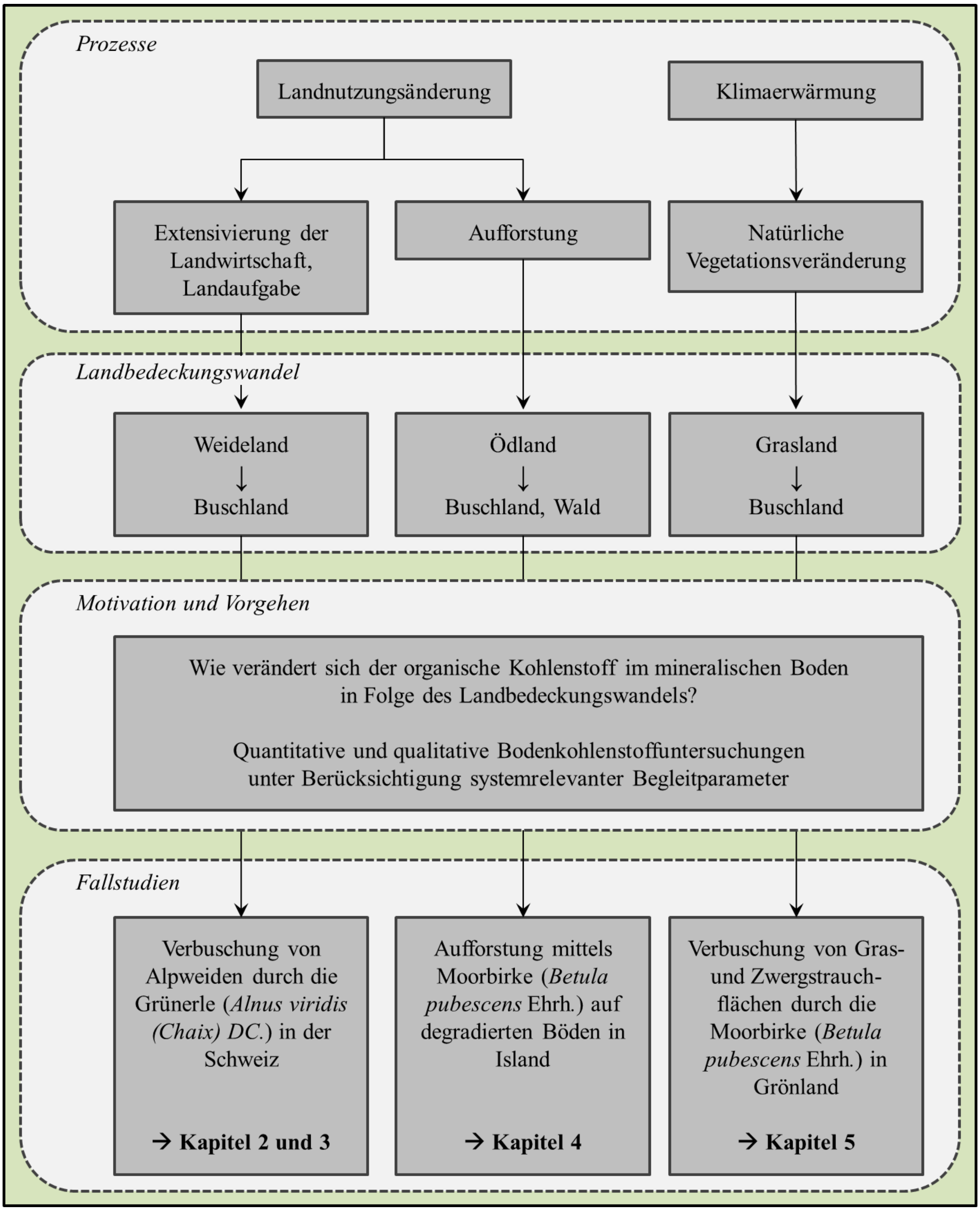

Abbildung 3: Das Konzept stellt den Aufbau des Dissertationsinhaltes dar und illustriert die Strukturierung der Arbeit.

1.7 Einführung in die drei Untersuchungsgebiete

Fallstudie I (Kapitel 2 und 3): Verbuschung von Alpweiden durch die Grünerle (*Alnus viridis* (Chaix) DC.) in der Schweiz

Die forst- und landwirtschaftliche Nutzung der subalpinen Flächen in den Gebirgen Europas hat die Bildung einer einzigartigen Landschaft mit hohen ökologischem Wert zur Folge (Gellrich and Zimmermann, 2007). Politische, soziale und wirtschaftliche Änderungen, welche anfangs des 20. Jahrhunderts einsetzten, haben jedoch zur Polarisierung der Landnutzungsintensität in diesen Gebieten geführt (Flury et al., 2013; Bätzing, 2015). Dabei ist mit Anreizen versucht worden, anstelle der lokalen Selbstversorgung die Produktion auf den globalen Markt auszurichten (MacDonald et al., 2000; Tasser et al., 2011). Doch die Topographie lässt den vorausgesetzten Maschineneinsatz nur bedingt zu und Boden sowie Klima limitieren eine Diversifizierung der Produkte und sind für eine kurze Vegetationszeit verantwortlich. Deshalb hat nebst der Intensivierung der Landwirtschaft auf den besser zugänglichen Flächen, welche sich meist in Tallagen und in der Nähe der Gehöfte befinden, die Aufgabe der schwieriger zugänglichen Nutzflächen in weiterer Entfernung und/oder Hängen mit grösserer Hangneigung stattgefunden (MacDonald et al., 2000; Tasser et al., 2011; Bätzing, 2015). Diese Umstrukturierungen haben zum Rückgang der Landwirtschaftsbetriebe geführt, welcher in den Schweizer Alpen zwischen 1985 und 2009 39 % betragen hat (BfS, 2013). Als Folge davon sind 372 km^2 aufgegeben und nicht mehr genutzt worden, was einem Rückgang von 34.5 % entspricht (Schubarth and Weibel, 2013).

Die in dieser Form stattfindende Landaufgabe wird als Hauptgrund für die Wiederbewaldung durch die Baum- und Buschvegetation in den Alpen betrachtet (Anthelme et al., 2002; Gehrig-Fasel et al., 2007; Gellrich et al., 2007; Cocca et al., 2012; Huber and Frehner, 2013). Dabei nimmt die Fläche der Buschvegetation stärker zu als jene der Baumvegetation. Die Buschfläche hat in den Schweizer Alpen zwischen dem ersten und vierten Forstinventar (1985-2013) um 113 km^2 (+21 %) zugenommen (WSL, 2015a, 2015b). Die Grünerle (*Alnus viridis (Chaix) DC = Alnus alnobetula (Ehrh.) K. Koch*) als eine sich schnell ausbreitende Pionierart, welche in Symbiose mit dem Bakterium *Frankia alni* (Stickstofffixierer) steht (Dawson, 2008), ist dabei die weitverbreiteste Buschart und wächst auf ca. 70 % der ausgewiesenen Buschfläche (Cioldi et al., 2010). Weiter ist die Bereitstellung von Pflanzenmaterial, das als Quelle für den organischen Bodenkohlenstoffvorrat dient, in Form der jährlichen Nettoprimärproduktion (Biomassezuwachs und Streuabfall) von *A. viridis* höher als bei anderen Baumarten oder von Pflanzenarten, welche auf subalpinen Alpweiden wachsen (Wiedmer and Senn-Irlet, 2006; FOEN, 2015; Bühlmann et al., 2016). Mit diesen Eigenschaften verfügt die Art *A. viridis* über die Möglichkeit, Böden von subalpinen Weideflächen während der Verbuschung signifikant zu verändern. So konnten Caviezel et al. (2014) die Veränderungen der physikalischen Bodeneigenschaften im Hinblick auf Bodenstabilität und Bodenerosion dokumentieren. Laut dem Waldgesetz der Schweiz werden Standorte von *A. viridis* als Waldfläche klassifiziert (Bundesgesetz über den Wald vom 4. Oktober 1991, Waldgesetz (WAG), SG 921.0). Ihr Verhalten hinsichtlich Wuchsform, Ausbreitungsverhalten und Produktivität unterscheidet sich jedoch von anderen Schweizer Waldbaumarten (Düggelin and Abegg, 2011; FOEN, 2014). Daher werden die Annahmen, dass sich der Bodenkohlenstoff (ca. 69 t C ha^{-1}) gleich verhält wie bei anderen Baumarten im subalpinen Raum, an dieser Stelle in Frage gestellt (FOEN, 2015).

Die Ziele der Fallstudie I (Kapitel 2 und 3) sind, das räumliche und zeitliche Ausbreitungsmuster von *A. viridis* im Unteralptal zu erfassen und die Dynamik des Bodenkohlenstoffes während der Verbuschung von subalpinen Alpweiden durch *A. viridis* zu dokumentieren.

Fallstudie II (Kapitel 4): Aufforstung mittels Moorbirke (*Betula pubescens* Ehrh. *ssp. czerepanovii*) auf degradierten Böden in Island

Birkenwälder kommen natürlicherweise in Island in den tieferen Lagen bis zu einer Höhe zwischen 200 bis 400 m ü. M. vor (Wielgolaski, 2005; Wöll, 2008). Schätzungen hinsichtlich der Gesamtbuschwaldfläche haben ergeben, dass vor der Besiedlung der nordatlantischen Insel durch die Wikinger um 871 n. Chr. ungefähr 25 % des Landes mit Buschwäldern von *Betula pubescens* Ehrh. bewachsen gewesen sind (Wöll, 2008). Dieses natürliche Ökosystem ist jedoch anfällig auf Störungen. Dafür ist auch der typische Bodentyp – der fruchtbare „Brown Andosol“ (Arnalds, 2004) – verantwortlich, welcher aufgrund der Mineralogie eine geringe Kohäsion aufweist und deshalb anfällig auf Wind- und Wassererosion ist (Aradottir and Arnalds, 2001; McDaniel et al., 2012). Die Begründung für die geringe Kohäsion liegt in der Tatsache, dass vulkanische Tonminerale kugelförmige Strukturen bilden und daher eine andere geometrische Gestalt aufweisen als bsw. Schichtsilikate (Arnalds, 2008, 2015a). Eine weitere Eigenschaft des Bodentyps ist seine hohe Wasserrückhaltekapazität. Jedoch hat sie bei Wasser gesättigtem Zustand in Kombination mit den erwähnten Kohäsionseigenschaften und einem Störungsereignis zur Folge, dass die Bodenmatrix in den thixotropischen Zustand übergehen kann, sich dadurch der Viskositätszustand des Bodens ändert und gravitative Massenbewegungen (z.B. Erdrutsche) als Erosionsformen ausgelöst werden können (McDaniel et al., 2012; Arnalds, 2015b). Im Hinblick auf die Kohlenstoffspeicherung weisen die Böden mit ihren vulkanischen Tonmineralien eine hohe spezifische Oberfläche auf (Ferrihydrite: 200 bis 500 m^2 g^{-1} und Allophane resp. Imogolite: 700 bis 1500 m^2 g^{-1}) (McDaniel et al., 2012).

Die Besiedlung der Insel durch *B. pubescens* Ehrh. und deren Ausbreitung hat im Boreal um ca. 8500 Jahre vor heute stattgefunden (Hallsdóttir, 1995). Seither ist das natürliche Ökosystem der Birkenbuschwälder Störungen durch vulkanische Aktivitäten ausgesetzt gewesen, wobei die Resilienz des Systems trotz teils starken Vulkanaschedepositionen erhalten geblieben ist und dadurch keine nachhaltige Schädigung des Vegetationssystems verursacht worden ist (Dugmore et al., 1995; Aradottir and Arnalds, 2001; Dugmore et al., 2009). Dies hat sich jedoch mit der Besiedlung Islands durch die Wikinger geändert und die Degradation der Buschwald- und anderer Ökosysteme (z.B. Grasland, Weidenbuschvegetation, Zwergstrauchheide) hat eingesetzt (Aradottir and Arnalds, 2001). Die Lebensweise, welche durch die Landwirtschaft mit Viehhaltung geprägt war, hat die Rodung grosser Gebiete für die Errichtung von Gehöften, Weide- und Heuflächen verlangt. Das Holz ist mittels Köhlerei zu Holzkohle weiterverarbeitet worden, die für die Eisengewinnung und zum Schleifen der Sensen genutzt worden ist (Arnalds, 1987). Zusätzlich zur direkten anthropogenen (Um-) Nutzung der Waldflächen hat die Übernutzung durch die Weidetiere (v.a. Schafe) zur weiteren Degradation des Ökosystems geführt. Nebst dem Vegetationswechsel durch das Fressverhalten hin zu weniger proteinhaltigen Pflanzengesellschaften und dem vermutlichen Abfressen der Stockausschläge, was die wichtigste Verjüngungsmethode von *B. pubescens* Ehrh. ist, haben vermehrt aufkommende vegetationslose Bodensenken für die Tiere als Schutzorte gedient, an welchen ebenfalls die Degradation und Erosion des Bodens begonnen hat (Aradóttir et al., 2001; Aradottir and Arnalds, 2001; Arnalds, 2015c). Währendem die Winderosion bei trockenen Bodenbedingungen gewirkt hat, hat die Wassererosion bei geringen Infiltrationsraten im Winter infolge von Bodenfrost oder bei Wassersättigung nach Niederschlägen stattgefunden (Orradottir et al., 2008; Arnalds, 2015b). Schätzung berechnen, dass seit der Besiedlung vor 1100 Jahren der jährliche Verlust an Bodenmaterial bis zu 30 Mt a^{-1} betragen haben kann (Arnalds, 2000; Óskarsson et al., 2004). Bezüglich des Bodenkohlenstoffes wird der Verlust an organischem Bodenkohlenstoff während dieser Zeit auf 120-500 Mt geschätzt (Arnalds, 2000; Óskarsson et al., 2004). Als Folge davon hat sich die Fläche der Wüstengebiete in Island von 5000 bis 15000 km^2 zum Zeitpunkt der Besiedlung auf ungefähr 50000 km^2 (inkl. den degradierten Flächen) vergrössert, was die Hälfte der Landesfläche ist (Arnalds, 2000). Die Erosions- und Akkumulationsprozesse haben zudem zur Bildung von Bodenprofilen geführt, die räumlich betrachtet sehr heterogen verteilt sein können. Gegenwärtig kann demnach das Bodenmaterial in den ersten Dezimetern unter der Geländeoberfläche aus einem freigelegten Paläoboden oder akkumuliertem Erosionsmaterial verschiedenster Herkunft und Materialeigenschaft bestehen (Dugmore et al., 2009).

Seit 1907 bekämpft die Isländische Bodenschutzbehörde (*Landgræðsla ríkisins*) die Wüstenbildung, indem sie mittels Sturmwällen aus Stein und Strandroggen (*Leymus arenarius*) die Wanderdünen zum Stillstand zwingt. Weiter leistet sie Aufklärungsarbeit bei den Landwirten, um die Überweidung zu stoppen. Im Herbst wird die Erodibilität der vegetationslosen Böden durch die Abdeckung mit Stroh gesenkt, um die Bodenerosionsraten während des Winters minimieren zu können. Und die Rekultivierung der erodierten Flächen wird mittels Düngereinsatz, Ansaat und Aufforstungsprogrammen durchführt (Runólfsson, 1987; Aradóttir, 2003; Arnalds, 2005; Croft et al., 2009; Runólfsson and Ágústsdóttir, 2010). Diese Rekultivierungs- und Aufforstungsanstrengungen werden seit den 1990-er Jahren zusätzlich vom Isländischen Staat finanziell unterstützt. Denn die stark degradierten Böden weisen ein hohes Kohlenstoffspeicherungspotential auf (Houghton and Goodale, 2004; Ministry for the

Environment, 2007; Lal, 2009; Hellsing et al., 2016). Die Begründungen für das hohe Speicherungspotential sind quantitativer und qualitativer Natur. In einem weltweiten Vergleich mit anderen durchlüfteten Bodentypen speichern Andosole am meisten Kohlenstoff (Batjes, 1996), weil die Mineralogie der Andosole sich für die Speicherung und Stabilisierung von Kohlenstoff durch die Bildung von Allophan-Humus- und Metall-Humus-Komplexen eignet (Arnalds, 2015d). Nach Arnalds (2008) werden in Island die degradierten und wüstenhaften Böden als Vitrisole klassifiziert. Ihr Vorkommen erstreckt sich mehrheitlich entlang des Isländischen Vulkangürtels (Arnalds et al., 2001; Arnalds and Óskarsson, 2009), wo das Material an der Geländeoberfläche mobil ist und Erosions- und Akkumulationsprozesse stattfinden (Arnalds, 2010). Das Bodenmaterial der Vitrisole weist wegen der Deposition von relativ schwach verwittertem Material und der fehlenden Vegetation, die als Säuredonator dienen würde, einen hohen pH-Wert auf (Arnalds, 2008). Wegen dem geringen Verwitterungsgrad ist der Anteil an vulkanischen Tonmineralien gering und das Fehlen der Vegetation resultiert in einer SOC Konzentration, welche unter 1 % C liegt (Arnalds, 2008). Im Gegensatz dazu weisen „Braune Andosole“ einen pH-Wert zwischen 5.5 und 6.5, eine SOC Konzentration zwischen 1 und 12 % und die höchsten Tonmineralgehalte der Isländischen Böden auf (Arnalds, 2008). Daher liegt der SOC Vorrat der Vitrisole (< 45 t C ha^{-1}) im Vergleich zu „Braunen Andosolen“ der produktiven Ökosysteme (227 t C ha^{-1}) deutlich tiefer (Óskarsson et al., 2004). Laut Lal (2009) können die Böden der Vitrisole infolge von Rekultivierungsmassnahmen als SOC Senken auf einer Fläche von 50000 km^2 in Betracht gezogen werden.

Die Einführung in das Thema der Fallstudie „Aufforstung“ zeigt deutlich die Einzigartigkeit der Isländischen Gegebenheiten in dieser Hinsicht auf. Unter diesen Umständen ist es besonders schwierig, geeignete Referenzwerte für das Kohlenstoffspeicherungspotential der Böden, welche nicht von Island selber stammen, im Zusammenhang mit der Aufforstung auf Isländischen Böden zu finden. Die Ziele der Fallstudie II sind daher die Quantifizierung und Beschreibung des Bodenkohlenstoffes, welcher durch die Aufforstung und die damit verbundene Landbedeckungsänderung zusätzlich gespeichert wird. Weiter soll das Speicherungspotential von aufgeforsteten Birkenbeständen in Bezug auf den Bodenkohlenstoff abgeschätzt werden. Die Fallstudie ist Teil des dreijährigen (2008-2011) *KolBjörk* Forschungsprojektes, das die ökologischen Auswirkungen der Wiederbewaldung durch *B. pubescens Ehrh.* im *Hekluskógar* untersucht (Halldórsson et al., 2011).

Fallstudie III (Kapitel 5): Verbuschung von Gras- und Zwergstrauchflächen durch die Moorbirke (*Betula pubescens Ehrh.*) in Grönland

In der zirkumpolar vorkommenden bioklimatischen Zone der Arktis ist die Klimaerwärmung während des 20. Jahrhunderts zwei- bis dreimal höher als der globale Durchschnitt gewesen (Field et al., 2013; Hansen et al., 2010). Für das 21. Jahrhundert wird eine weitere Zunahme der Globaltemperatur von 1.4 bis 5.8 °C prognostiziert (Field et al., 2013). Damit verlängert sich die

Zeit der Eis- und Schneeschmelzen sowie der Vegetationszeit, was das Auftauen des Permafrostes, das Waldbrandrisiko, die anthropogene Nutzung und die Änderung der Vegetationszusammensetzung fördert (Callaghan et al., 2004). Satellitengestützte Untersuchungen der zirkumpolaren Vegetationseigenschaften haben bereits eine Zunahme des normalisierten differenzierten Vegetationsindexes (NDVI) festgestellt, was auf eine Vegetationsveränderung hin zu mehr chlorophylbildenden Pflanzenarten hindeutet. Dabei breitet sich die Tundravegetation in den Bereich der polaren Wüste aus (Bhatt et al., 2010; Jeong et al., 2012). Und im südlichen Bereich der Tundra wachsen Arten der borealen Vegetationszone ein (Payette et al., 2001; Myers-Smith et al., 2011). Modelle prognostizieren einen Rückgang der Tundrafläche zugunsten jener des Boreals um 11 bis 55 % bis 2100 (Pearson et al., 2013; Zhang et al., 2013), weil das Grenzökoton zwischen der Zwergstrauchtundra und der Waldtundra am schnellsten auf diese Umweltänderungen reagieren wird (Myers-Smith et al., 2011).

Definitionsgemäss bildet die Baumgrenze die Trennung zwischen Tundra und Boreal (Walker et al., 2005). Diese Grenze ist jedoch nicht eindeutig erkennbar, denn die Tundra lässt sich in mehrere Subzonen unterteilen, was in einer räumlichen Sukzessionsreihe generell in Richtung der Zone des Boreals resultiert (Walker et al., 2005). Der Atlas der zirkumpolaren, arktischen Vegetation unterteilt daher die Tundra in fünf physiognomische Hauptkategorien: Nördliche Polarwüste mit u.a. Kryptogamen und Polsterpflanzen, Grastundra, Tundra mit lateral wachsender Zwergstrauchvegetation, Tundra mit emporwachsender Zwergstrauchvegetation und Feuchtgebiete (CAVM Team, 2003). Dabei wird die Zwergstrauchvegetation mit einer Wuchshöhe bis zu 0.4 m definiert (CAVM Team, 2003; Myers-Smith et al., 2011). Die Waldtundra, welche aus einzel- oder mehrstämmigen Büschen oder Bäumen zwischen 0.4 und 4.0 m besteht (Myers-Smith et al., 2011), schliesst in der Sukzession der Zwergstrauchvegetation an. Die Waldtundra unterscheidet sich aufgrund der Wuchsdichte und den dominierenden Arten vom borealen Nadelwald (FAO, 2012).

Die Zonierung ist in Realität jedoch komplexer. Brandereignisse, Bodenfeuchtigkeitsverhältnisse, Topographie und Substrateigenschaften verursachen ein mosaikartiges Muster der oben erwähnten Vegetationszonen und lassen die Sukzessionsreihen auch kleinräumig oder entlang eines Höhengradienten auftreten (Tape et al., 2006; Petrenko et al., 2016; Henkner et al., 2016; ACIA, 2004). Aufgrund dieser mosaikartigen Musterung hat die Verbuschung der Tundra nach Myers-Smith et al. (2011) drei Formen: das Zusammenwachsen bereits bestehender Buschverbände, die Zunahme der Wuchshöhe durch die Veränderung der Wuchsform und die laterale und vertikale Verschiebung der Baumgrenze durch die Kolonialisierung von Gebieten ausserhalb der aktuellen ökologischen Nische.

Grönland, die grösste Insel der Erde, liegt zwischen 59°46'N und 83°04'N (Quinn and Woodward, 2015), weshalb sich alle drei bioklimatischen Zonen der Arktis (Hoch-, Nieder und Subarktis) finden lassen (Daniëls et al., 2014). Das zuvor beschriebene Grenzökoton der

Waldtundra befindet sich im Südwesten Grönlands, wobei die aufrecht wachsenden Birken (*Betula pubescens*) in windgeschützten Gunststandorten im Innern der Fjorde wachsen (Polunin, 1938; Böcher, 1979; Feilberg, 1984). In diesen Bereichen wird das Klima als sub-arktisch und sub-kontinental beschrieben (Böcher, 1979; Hanna and Cappelen, 2002). Seit Messbeginn von 1881 sind die Temperaturen für die Zeitperiode 2001-2012 in diesem Gebiet im Winter um 4.4 °C und im Sommer um 1.8 °C höher gewesen als im Referenzzeitraum von 1881-1910 (Hanna et al., 2012). Für das 21. Jahrhundert sagen Szenarien zudem eine Temperaturerhöhung von 3.3 °C, eine verlängerte Vegetationszeit von zwei Monaten und eine Verschiebung der sub-arktischen Buschvegetation voraus (Masson-Delmotte et al., 2012; Normand et al., 2013; Christensen et al., 2016). Studien zur Vegetationsdynamik zeigen jedoch, dass es in den vergangenen Jahrzehnten kaum zu Landbedeckungsänderungen entlang der Grönländischen Küsten gekommen ist, welche eindeutig auf die Klimaerwärmung zurückzuführen sind (Drees and Daniëls, 2009; Callaghan et al., 2011; Daniëls and de Molenaar, 2011; Damgaard et al., 2016).

Bezüglich Bodenkohlenstoffuntersuchungen sind erst in den vergangenen Jahren im Vorland des Ørkendalen Gletschers (Westgrönland) Untersuchungen durchgeführt worden (Henkner et al., 2016). Die Ergebnisse zeigen, dass die Vegetationsbedeckung, die im Gletschervorland aus Gräsern und Zwergstrauchheide bestanden hat, als Hauptfaktor für die Menge an Kohlenstoff im Boden betrachtet werden kann (Henkner et al., 2016). Ähnliche Untersuchungen fehlen jedoch für das sub-arktische Grenzökoton in Südwest-Grönland. Ein weiterer Grund für die Untersuchung ist die physiognomische Eigenschaft des Grenzökotons, das sich nicht eindeutig einem der beiden Biome des Boreals oder der Tundra zuteilen lässt. Untersuchungen konzentrieren sich auf typische Vegetations- und Bodengesellschaften der beiden Biome (Jobbágy and Jackson, 2000).

In Kapitel 5 zeigt die Fallstudie III deshalb auf, wie sich gegenwärtig die Bodenkohlenstoffvorräte unter verschiedenen Vegetationstypen im Waldtundra-Grenzökoton in der Nähe von Igaliku verhalten. Darauf aufbauend wird diskutiert, unter welchen Bedingungen sich die Buschvegetation ausbreiten kann und welche Auswirkungen eine Landbedeckungsänderung hin zu mehr Buschvegetation auf den Bodenkohlenstoff in quantitativer und qualitativer Hinsicht haben kann.

KAPITEL 2

Green alder encroachment in the European Alps: The need for analyzing the spread of a native-invasive species across spatial data

Momentaufnahme der floristischen Lebensräume im Bereich des „Hinter Älpetlital" im Unteralptal mit dem Gemsstock (2962 m ü. M.) in der Mitte im Hintergrund. Zu sehen sind Grünerlengebüsche, Hochstaudenflure und Bergfettweiden entlang des Osthangs des Tals und auf einem in historischer Zeit geformten Schwemmfächer. Aufgenommen von M. Hunziker am 21. Juli 2011.

Green alder encroachment in the European Alps: The need for analyzing the spread of a native-invasive species across spatial data

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Abstract

Forest regrowth is an ongoing process in the European Alps. In the Unteralptal, a valley in central Switzerland, landscape characteristics show considerable change towards the encroachment of green alder (Alnus viridis (Chaix) DC.). Initially, green alder was described to grow on moist, north-facing and steep slopes of high geomorphic activity. However, the recent spreading of green alder described in several studies over the alpine arc, questions the ecological habitat described in historic literature. Thus, a time series of aerial photographs and a digital elevation model (DEM) with a resolution of 2 m were used to find proxies for green alder encroachment. The cover of “new shrub areas” was analyzed based on relief parameter values and geomorphic landforms. The results show that green alder is spreading on more gentle slopes and well-drained areas, as well as on areas with lower geomorphic activity than anticipated. Thus, the habitat spectrum of green alder is much wider than assumed and encroachment has potentially greater consequences for landscape ecology than expected.

Keywords: subalpine pastures, land abandonment, land cover change, green alder encroachment, relief parameter, landform analysis

2.1 Introduction

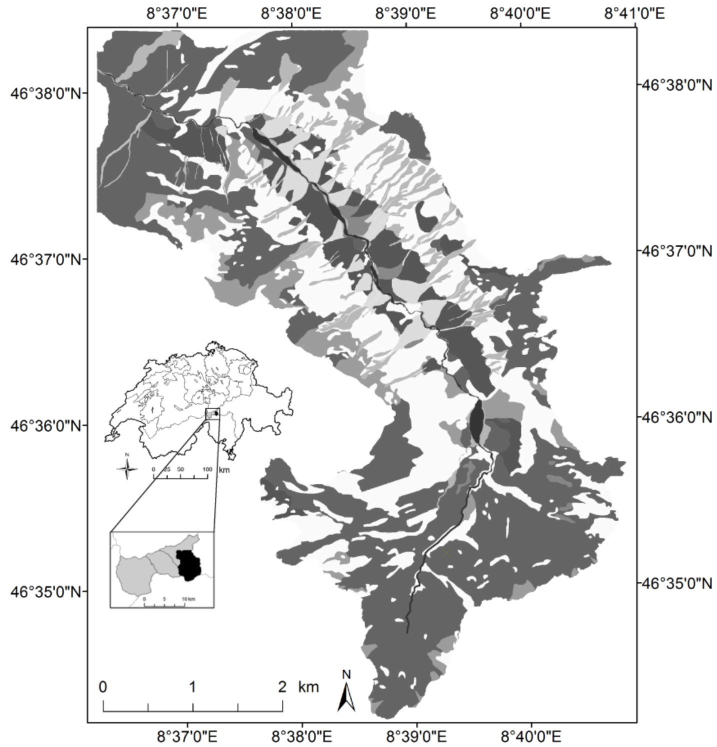
While the tropics experience a pervasive deforestation due to the increase of commercial agriculture and timber harvesting (FAO, 2010), semi-arid and arid biomes (Eldridge et al., 2011) as well as several mountain ranges in developed countries experience an expansion of woody species (MacDonald et al., 2000; Alewell and Bebi, 2011). In contrast to the increase of woody vegetation in semi-arid and arid environments that is driven by overgrazing, fire suppression and climate change (D'Odorico et al., 2012), the encroachment of shrubs and trees in the mountain ranges of developed countries is mainly caused by a marginalization of agricultural areas (MacDonald et al., 2000). In Europe, mountain ranges which experienced an expansion of woody flora during the twentieth century include the Alps (Gellrich et al., 2007; Tasser et al., 2011), the Pyrenees (Mottet et al., 2006; Montané et al., 2007), and the Carpathians (Wiezik et al., 2013). The forested area in Switzerland increased by 1304 km^2, including 174 km^2 shrub woodland, between the observations in 1983/85 and 2009/11 (WSL, 2012a, 2012b, 2012c, 2012d). Almost all of the newly forested area (97.5%) lies within the Swiss Alpine region (Brändli, 2010). In the Swiss Alps, green alder (*Alnus viridis* (Chaix) DC*. = Alnus alnobetula* (Ehrh.) K. Koch), an early successional species, is a major component of the increasing subalpine shrub woodland: 70% of the shrub areas consist of green alder, 20% of dwarf mountain pine (*Pinus mugo* subsp. *prostrata*)*,* while the remaining 10% is dominated by hazel (*Corylus avellana*) and various willow species (*Salix sp.*) (Brändli, 2010). The high percentage of green alder among the re-growing woody flora can be explained by its strong colonization ability and high seed production (Farmer et al., 1985). Further, green alder shows competitive abilities due to its "horizontal competition strategy " described by Mallik et al. (1997). The relatively short height of individuals (4 m) and the high density of stems and foliage can impede the growth of vertical competition strategist such as native conifers. Historically, the ecological requirements of green alder are described as naturally restricted to steep, north-facing, moist slopes and disturbed habitats with high geomorphic activity, such as avalanche and debris flow tracks and channels in the subalpine belt (Schröter, 1908; Richard, 1969; Hörsch, 2003). However, a growing number of studies show that green alder spread on recently disused subalpine pastures outside those historically defined habitat properties (Anthelme et al., 2003; Wiedmer and Senn-Irlet, 2006; Camacho et al., 2008). Apparently, the initially described habitat does not coincide with the recent spreading observed in several studies over the Swiss (Wiedmer and Senn-Irlet, 2006; Huber and Frehner, 2013) and the northern French Alps (Anthelme et al., 2001, 2007; Camacho et al., 2008). The wider expansion of green alder raises the question whether the assumed ecological restrictions actually apply and highlights the need to improve the understanding of green alder spreading (Eggenberg, 2002). The explicit spatial analysis of areas that experienced an encroachment of green alder can provide a further insight in the ecological properties of new colonized areas. However, a recent spatial analysis of the habitats colonized by the green alder is lacking.

Thus, the spatial analysis is of great importance as the rugged topography in mountain environments leads to a varied spatial distribution of radiation, air and soil temperature and soil moisture (Hörsch, 2003), as well as soil coarse fraction, bulk density and organic carbon (Hoffmann et al., 2014a, 2014b). This results in a high diversity of micro-ecological habitats. The relationship between topography and vegetation composition has been shown in several studies (Barrio et al., 1997; Pinder et al., 1997; Franklin, 1998; Guisan et al., 1998; Hörsch et al., 2002; Hörsch, 2003; Pfeffer et al., 2003). Topography can be characterized by geomorphometric relief parameters, such as slope angle, slope exposition, elevation and curvature, each derived from digital elevation models (DEM) on the scale of single pixels. Thus, the explicit spatial analysis using DEMs and its relief derivatives provides proxies for habitat conditions indicative of a specific vegetation composition (Hörsch, 2003). Besides topography, vegetation associations are also controlled by surface stability and geomorphic activity (Swanson et al., 1988). Landforms reflect surface stability and geomorphic activity as their development is controlled by the frequency of the associated surface processes (Renschler et al., 2007). This study therefore also examined the changes in green alder cover related to landforms, which act as proxies for geomorphic activity. Consequently, the study addresses the following research questions: i) Which relief parameter values characterize the area covered by green alder? ii) Does geomorphic activity characterize the area covered by green alder? iii) Is there evidence for alternative factors controlling green alder cover? To answer these specific research questions, the Unteralp valley (Unteralptal) was chosen as study site because its vegetation dynamics (Caviezel et al., 2014; Hunziker et al., 2017) and land use changes (Caviezel et al., 2010) are typical for large parts of the Alps (MacDonald et al., 2000; Wettstein, 2001; Wiedmer and Senn-Irlet, 2006; Bätzing, 2015). Thus, the study enables a general conclusion, whether the ecological habitat described in the historic literature actually applies to alpine areas experiencing shrub encroachment.

2.2 Methods

2.2.1 Study site description

The Unteralptal, a side valley of the main Urserntal in central Switzerland (46.37°N, 8.38°E), comprises an area of 35 km^2 stretching from the highest point at 2900 m a.s.l. to the mouth of the main Urserntal near the village of Andermatt at 1442 m a.s.l. (Ambühl et al., 2008). Geologically, the Unteralptal is part of the Gotthardmassiv and consists of paragneiss, migmatit and orthogneiss (Ambühl et al., 2008). The climate is alpine with a mean air temperature of 3.4°C and a mean annual rainfall (1961–1990) of 1422 mm per year at the nearby MeteoSwiss climate station in Andermatt (1442 m a.s.l.). Based on the FAO World Reference Base for soil resources (FAO, 2006), the dominant soil types in the catchment are Leptosols on steep valley slopes and Podzdocambisols and Cambisols on lower slopes. The predominant soil texture is silty sand and sandy silt. The previously glaciated valley is about 10 km long, drained by the river Unteralpreuss and characterized by a rugged terrain. The steep slopes of the U-shaped valley lead to high topographic energy and geomorphic activity resulting in diverse erosional and depositional landforms as shown in Abbildung 4. The main vegetation types nowadays are alpine grasslands, dwarf-shrubs, mostly alpine rose (*Rhododendron ferrugineum*) and bilberry (*Vaccinium myrtillus*), and shrubs, mostly green alder (*Alnus viridis* (Chaix) DC.) with single mountain ashes (*Sorbus aucuparia*).



	True surface area [ha]	Percentage of total study area
Valley bottom		
Unteralpreuss river		
Alluvium	32.8	1.3
Orographic left side		
Bedrock	401.8	16.2
Channel	45.7	1.8
Debris cone	1.2	0.1
Debris flow track	9.0	0.4
Moraine depostit	374.9	15.1
Rockfall deposit	117.5	4.7
Talus cone	30.2	1.2
Talus sheet	87.7	3.5
Wetland	2.1	0.1
Orographic right side		
Bedrock	398.7	16.1
Channel	116.8	4.7
Debris cone	10.6	0.4
Debris flow track	9.3	0.4
Moraine depostit	596.2	24.1
Rockfall deposit	80.3	3.2
Talus cone	44.4	1.8
Talus sheet	110.6	4.5
Wetland	3.9	0.2

Abbildung 4: Study site and landform classification of the Unteralptal, the classification is based on the DEM2 hillshade, the geomorphologic map of the area (1:25000) and the aerial photographs (Messenzehl et al., 2014); terminology is used according to Ambühl et al. (2008). Further on, landforms have been allocated according to their positioning into orographic left and orographic right areas, as the orographic right side areas are characterized by a better accessibility by the drivable road.

Almost the total of the study area belongs to the Korporation Ursern, a cooperation of associated citizens under public law, *i.e.* communal land. Cooperative pasture use began in the 13th century. Due to grassland farming for centuries, the current vegetation shows strong anthropogenic influences. According to Rebsamen (1919) and Kägi (1973), there has once been a closed forest in the Urserntal and its side valleys, reaching to the through shoulder. The presence of conifer forest is proofed by documentary evidence, field names connected to forests, as well as by numerous findings of preserved trunks (Kägi, 1973). The forest has been cleared to gain pasture area and for firewood collection and the clearance is dated for 12th and 13th century (Kägi, 1973). Only small areas of conifer forest remained. Traditionally land use in the Urserntal was temporally and spatially restricted to avoid overgrazing and to preserve the quality of the pastures (Caviezel et al., 2010). The regulations led to a small scaled heterogeneous land use where the type of land use was adapted to natural site factors. The use restrictions were abolished in the beginning of the 1970s and pasture areas were merged to larger areas. Further observed changes include i) the disappearance of goat pastures and decrease of goat number as goats lost their economic importance and ii) the abandonment of steep areas that are difficult to access and have formerly been mown by hand and iii) the ceasing of firewood collection by the introduction of oil heating (Caviezel et al., 2010; Wunderli, 2010).The study region has been classified as region with a marginal decrease of agricultural area (Tappeiner, 2003). Thus, it is not considered as a region that is highly affected by land use change like for example the regions of the Piedmont and the Savoy with land abandonment rates of 72% (Tappeiner, 2003). However, the study region shows the overall trend in farming of most alpine mountain areas. Centrally located areas experienced an intensification, while remote areas unsuitable for a mechanization of farming experience a marginalization (MacDonald et al., 2000; Bätzing, 2015).

2.2.2 Derivation of land cover classification

Land cover types were identified by visually digitizing the area from aerial photographs (Swisstopo, Federal Office for Topography). The oldest available aerial photograph dates back to the year 1959. In order to calculate land cover change, a time series with images from 1959, 1979 and 2007 was generated. The aerial photographs were chosen based on availability, image quality, minimum shading and appropriate interval between the flight dates. The photographs from 1959 and 1979 are black and white images, the photograph of 2007 is an RGB (red, green, blue) image. The images were georeferenced and orthorectified, enabling the comparison changing vegetation compositions between different years. The geometrical corrections were performed by the ENVI software package (ENVI 4.7) using between 15 and 20 ground control points based on the digital topographic map, the digital elevation model (DEM2 and DEM25 above 2000 m a.s.l.) (Swisstopo, Federal Office for Topography) and the camera calibration protocols (Swisstopo, Federal Office for Topography). Due to the differences in illumination, in particular shading by surrounding mountains, it was not possible to perform digital image classification based on color differentiation. In some completely shaded areas, vegetation was not identifiable and those areas were therefore digitized on each aerial photograph, merged and excluded for calculating vegetation and vegetation change. The high spatial resolution of the aerial photographs made it possible to distinguish between four vegetation categories: i) green alder (*Alnus viridis* (Chaix) DC.) associations with closed canopy (subsequently called closed

Alnus viridis associations) (C *AV*), ii) green alder associations with a cover between 20% and 40% (Braun-Blanquet, 1964) (subsequently called open *Alnus viridis* associations) (O *AV*), iii) alpine rose (*Rhododendron ferrugineum*) and bilberry (*Vaccinium myrtillus*) associations (subsequently called dwarf shrubs associations) (D S), and iv) grassland. Vegetation mapping has been limited to areas below 2400 m a.s.l. because the maximum elevation of a single standing green alder shrub was determined at 2390 m a.s.l. The investigation area was thereby reduced to approximately 20 km^2. Grassland was calculated as “investigation area” minus “C AV-area” minus “O AV-area” minus “D S-area” minus “permanent areas” namely lakes, water currents, debris, rocks, streets and shaded areas. Ground truthing of vegetation composition and extent derived from the aerial photographs was performed in summer 2010.

2.2.3 Estimation of the landscape surface area

Conventionally, land cover is calculated using the planimetric area resulting from the projection of an uneven surface onto a 2-dimensional grid. The estimation of the area is most accurate for flat landscapes. However, the bias increases with topographic roughness (Zhiming et al., 2012). Therefore, the planimetric approach is not considered as sufficiently accurate to quantify land cover change in mountain areas with rough topography (Zhiming et al., 2012). Especially with regards to the calculation of fluxes or changes in stock, as for example carbon, or biomass calculations an exact quantification of the area is essential. Thus, a method to estimate the surface area of a pixel cell representing steep sloping terrain was applied. This technique is based on a moving 3×3 cells window algorithm and estimates the surface area cell-by-cell by using a triangulation method. Further information on the algorithm is given in Jenness (2004) and Zhiming et al. (2012). Prior to the land cover change analysis, the two approaches were compared. As the planimetric approach led to a significant underestimation of closed and open *Alnus viridis* associations by about 30% (Tabelle 1), the areas generated by the surface approach were subsequently used in this study. Some vegetated areas experienced a change “opposite to succession”, (*e.g.* converted from shrub to dwarf shrub or grass). Opposite succession mainly occurred as a consequence of ski slope construction, but also due to changes of the morphology of river bed and mud flow channels. Results of the area affected by vegetation change are presented as net changes for the time periods of 1959 to 1979 and 1979 to 2007. Green alder cover is calculated by combining the area encroached by closed (C *AV*) and open green alder associations (O *AV*). Results are shown as relative and absolute net change of green alder surface cover within a class.

2.2.4 Selection and derivation of primary and secondary relief parameters using DEM data

Prior to all the analysis of the relief parameters and the landforms of the study area, the DEM with a resolution of 2 m (DEM2) was adapted to address the research question. Therefore, a mean filter of the size of 3 × 3 cells was applied on the DEM without reducing the spatial resolution of the DEM. The pre-processing of the DEM attenuated micro-scale structures such as individual boulders. The processing was performed as such micro structured elements are most likely not relevant for the growth of green alder shrubs and could falsify information on slope or aspect of single pixels. In order to perform descriptive analysis of relief parameters for the areas that experienced shrub encroachment, pixel-based information of primary (slope angle, slope exposition) and secondary (solar radiation and topographic wetness index (TWI)) relief parameters was extracted from DEM2 and classified using ESRI ArcGIS (10.0). The pixel-based analysis of primary relief parameters of the areas with shrub encroachment provides quantitative information on their environmental site conditions. The landform classification was performed to provide additional information relating green alder encroachment to geomorphic activity. The landforms were further subdivided into "orographically left" and "orographically right" as the aerial photographs showed apparent differences in its green alder cover depending on the landform allocation in the valley. Tabelle 2 provides an overview on the analyzed parameters, their classification, derivation and their relevance for controlling habitat conditions and vegetation distribution.

Tabelle 1: Areas of the analyzed land cover categories calculated by the planimetric and surface technique for 1959 (below 2400 m a.s.l.).

Land cover categories	Planimetric area	Surface area	Underestimation by the planimetric analysis
	[ha]	[ha]	[%]
Shadow	117.3	174.1	48.4
River bed	21.8	22.2	1.6
Road	2.3	2.3	2.3
Closed *Alnus viridis* associations (C *AV*)	90.2	118.5	31.4
Open *Alnus viridis* associations (O *AV*)	14.0	18.1	29.1
Dwarf shrub associations (D S)	21.1	25.3	19.7
Grassland	1560.1	1792.8	14.9
Lake	3.3	3.3	0.0
Rock surface	124.3	193.6	55.7
Debris	114.4	125.1	9.4
Total	2068.9	2475.3	19.6

Tabelle 2: Analyzed geomorphometric and geomorphic parameters and their relevance for controlling habitat conditions and vegetation distribution.

	Parameter	Computed by a combination of	Classification	Relevance for controlling habitat conditions and vegetation distribution	Scale
Primary relief parameter	Slope inclination	First derivate of DEM 2	Literature (Sponagel et al., 2005); land use recommendations (Surber, 1973); geomorphic activity (Summerfield, 1999).	Frequency and intensity of gravitational geomorphic processes; influencing land use activity and intensity	Pixel-based 2 m × 2 m, in accordance to Dikau (1988), classified as “picorelief” to“microrelief A"
	Slope aspect	First derivate of DEM 2	Equal interval	Radiation; temperature; snow cover; influencing land use, the type of land use activity and intensity	Pixel-based 2 m × 2 m, in accordance to Dikau (1988), classified as “picorelief” to“microrelief A"
Secondary relief parameter	Topographic wetness index	Secondary derivate of DEM 2, computed by distributing water proportionally to the slope angle of neighboring pixels.	Geometric interval. Higher TWI values represent drainage depressions; lower values represent crests and ridges.	Soil moisture; channels and ravines.	Pixel-based 2 m × 2 m, in accordance to Dikau (1988), classified as “picorelief” to“microrelief A"
	Solar radiation	Secondary derivate of DEM 2; elevation, slope exposition inclination and topographic shading, accumulated for an approximated vegetation period of May 15 until Oct 15 (Schröter, 1908).	Equal interval	Radiation; soil moisture; temperature	Pixel-based 2 m × 2 m, in accordance to Dikau (1988), classified as “picorelief” to“microrelief A"

Fortsetzung

	Parameter	Computed by a combination of	Classification	Relevance for controlling habitat conditions and vegetation distribution	Scale
Landform classification	Bedrock Channel Debris cone Debris flow track Moraine deposit Rockfall deposit Talus cone Talus sheet Wetland	Visually, based on identifiable landforms from the DEM 2 hillshade; the geomorphologic map of the area (1:25000), the aerial photographs and ground truthing (Messenzehl et al., 2014)	Terminology based on Ambühl et al. (2008). Landforms have been allocated according to their positioning, into orographic left and orographic right side of the valley	Frequency and intensity of gravitational geomorphic processes; boundary of specific habitat conditions within a landform (interaction between several relief parameters); influencing land use activity and intensity	Geomorphic landforms in accordance to Dikau (1988), classified as “microrelief B” to “mesorelief A”

2.2.5 Analysis of the correlation between vegetation change, geomorphometric and geomorphic parameters

Pixel-based analysis of the areas with a net vegetation change for the periods 1959/1979 and 1979/2007 were combined with the primary and secondary relief parameters. This was performed by cross-tabulating the vegetation associations according to the defined relief parameter classes by using ArcGIS. The process was repeated with the vegetation associations and the classified landform types. The results of the pixel-based and the landform analysis are presented as absolute values, as well as relative increases in relation to the area of the corresponding relief class or the *landform* class. The absolute and relative results were obtained by combining the area encroached by open (O *AV*) and closed (C *AV*) *Alnus viridis* associations between 1959, 1979 and 2007.

2.3 Results and discussion

2.3.1 Green alder encroachment beyond its assumed ecological niche

Net changes of the different vegetation associations for the whole analyzed time span show that closed *Alnus viridis* (C *AV*) associations increased by 43% and open *Alnus viridis* (O *AV*) associations increased by 195% (Tabelle 3). The increase rates for closed and open *Alnus viridis* associations were higher for the time period of 1959 to 1979 (C *AV* + 25.6%, O *AV* + 123.3%) compared to the time period of 1979 and 2007 (C *AV* + 14.1% O *AV* +32.1%). The area in the Unteralptal that experienced either an encroachment by closed or open *Alnus viridis* associations (C *AV* + O *AV)* within 48 years amounts to 86.7 ha, corresponding to an increase of 63%.

Comparing the area currently covered by green alder (aerial photograph of 2007) to the area that corresponds to the ecological requirements of green alder described in literature, obvious discrepancies emerge (Abbildung 5). Out of 400 ha corresponding to the ecological requirements of green alder traditionally reported in the literature, only 73 ha show green alder cover. On the other hand, an area of 150 ha, which does not correspond to the mentioned ecological requirements, is overgrown by green alder. The fact that 67% of the land covered by green alder is not part of the initial habitat description illustrates that the shrub expands far beyond its "historical" ecological habitat. Thus, the assumed ecological habitat of green alder does not apply to the land cover in the Unteralptal.

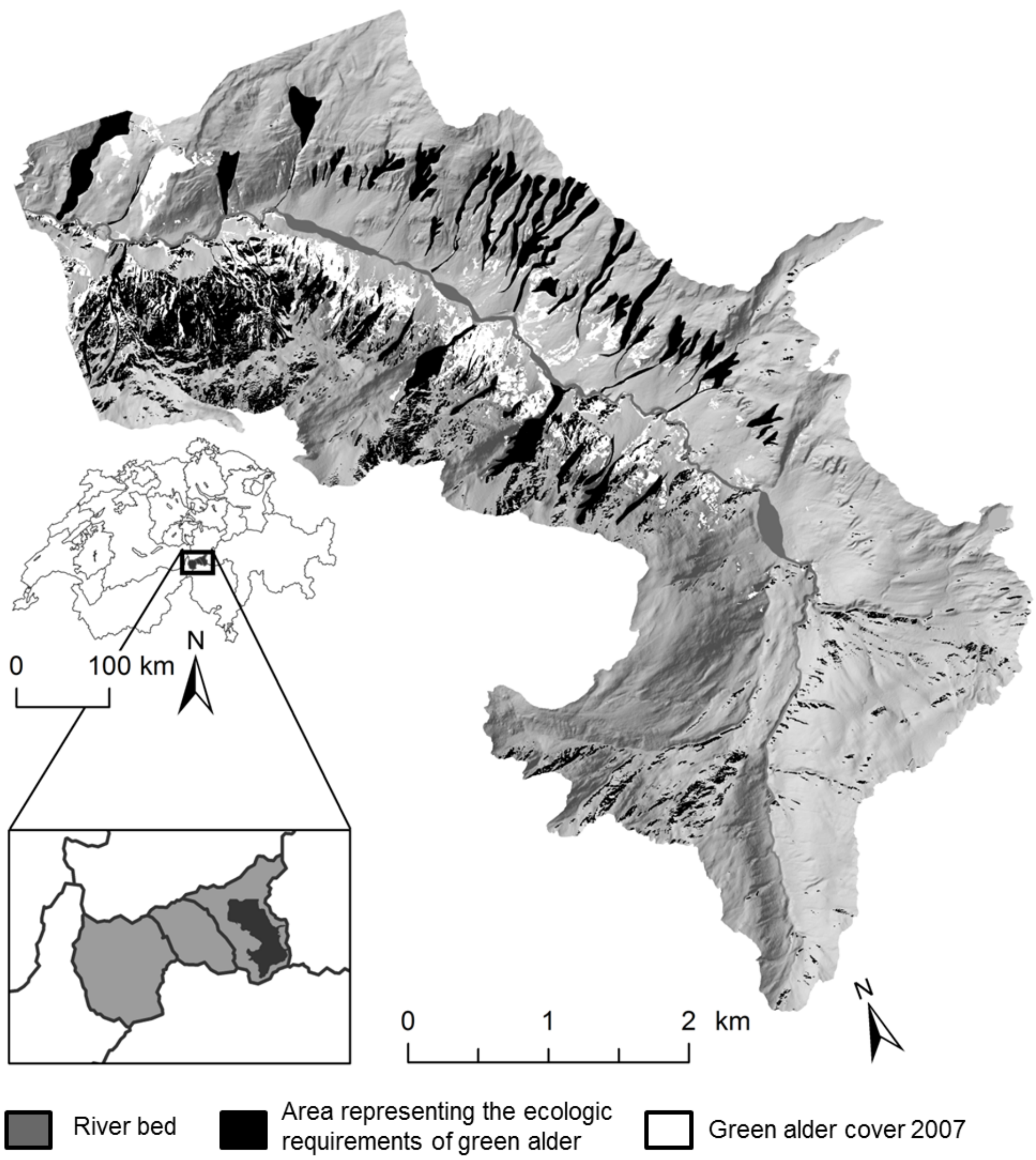


Abbildung 5: Discrepancy between the potential area (400 ha) for green alder growth (black) according to its ecologic requirements and the effective area (223 ha) of green alder growth in 2007. The size of the area (white) for bushes growing outside of the potential area was 150 ha. Potential areas were generated by conditional inquiries on the derivates of the DEM2 and the digitized landform classification in ArcGIS. The areas representing the ecologic requirements of green alder are north exposed with a minimum slope angle of 60% or are classified as either channels or debris flow tracks by not taking into account their slope angles or exposures (Schröter, 1908; Richard, 1969; Hörsch, 2003).

2.3.2 Geomorphometric proxies defining the ecologic niche of green alder

A factor often described as controlling green alder growth is slope angle. Slope angle can control geomorphic activity (Summerfield, 1999), but also indicate the kind and intensity of land use (Surber, 1973; Sutter and Keller, 2009). The analysis of slope angle (Abbildung 6a) revealed that steep slopes (>60%) still show the greatest proportional (18%) and absolute (168 ha) cover in 2007. However, increase rates between 1959 and 2007 are higher on slopes of 50-60% angle

(5.5%, corresponding to 15.7 ha). Green alder cover within the slope angle class of 34-50% also show remarkable increase rates. According to Surber (1973), sheep and goats are recommended to graze on slopes up to 60%, while cows can graze up to 50%. The increase of green alder cover on slopes within the angle classes of 50-60% and 34-50%, gives rise to the assumption that shrub encroachment can be linked to the documented decline of goats and to land use extensification in general (Caviezel et al., 2010; Wunderli, 2010, 2011).

Slope exposition analysis (Abbildung 6b) shows that green alder still reaches its maximum cover (26.8% of the total vegetated area within the category) at north-facing slopes. Also the increase of green alder cover is greatest at north- (6.5%) and northeast-facing slopes (6.6%). This underlines the preference of green alder for moist, north facing slopes described by Schröter (1908) and Richard (1969). However, the absolute increase (19.4 ha) at southwest-facing slopes is comparable to the increase at north- (22 ha) and northeast-facing (15.3 ha) slopes. Also, the area at south- (5.5 ha) and southeast- (2.6 ha) facing slopes increased. Thus, green alder form established stands at south-, southwest- and southeast-facing slopes which experience higher solar radiation and thus are characterized by drier habitat conditions. This result coincides with the outcome of a study in the eastern part of Switzerland by Huber and Frehner (Huber and Frehner, 2012, 2013), who also found an increase of green alder cover and southwest- and southeast facing slopes.

Tabelle 3: Net increase of the surface area for the vegetation categories “Closed *Alnus viridis* associations” (C AV), “Open *Alnus viridis* associations” (O AV), “Dwarf shrub associations” (D S) and grassland between 1959, 1979 and 2007 (below 2400 m a.s.l.).

Vegetation categories	Total vegetated area 1959	Change between 1959 and 1979		Total vegetated area 1979	Change between 1979 and 2007		Total vegetated area 2007	Change between 1959 and 2007	
	[ha]	[ha]	[%]	[ha]	[ha]	[%]	[ha]	[ha]	[%]
C AV	118.5	30.3	25.6	148.9	21.0	14.1	169.9*	51.4	43.3
O AV	18.1	22.3	123.3	40.4	13.0	32.1	53.3	35.3	195.0
D S	25.3	1.0	4.0	26.3	-4.3	-16.3	22.0	-3.3	-12.9
Grassland	1792.8	-57.7	-3.2	1735.1	-25.5	-1.5	1709.7	-83.2	-4.6
Total vegetated area	1954.7*			1950.7*			1954.9*		

*The small inaccuracies in calculated numbers for the total vegetated area for the different years, as well as for vegetation dynamics, based on changes in the riverbed area, road construction and rounding errors.

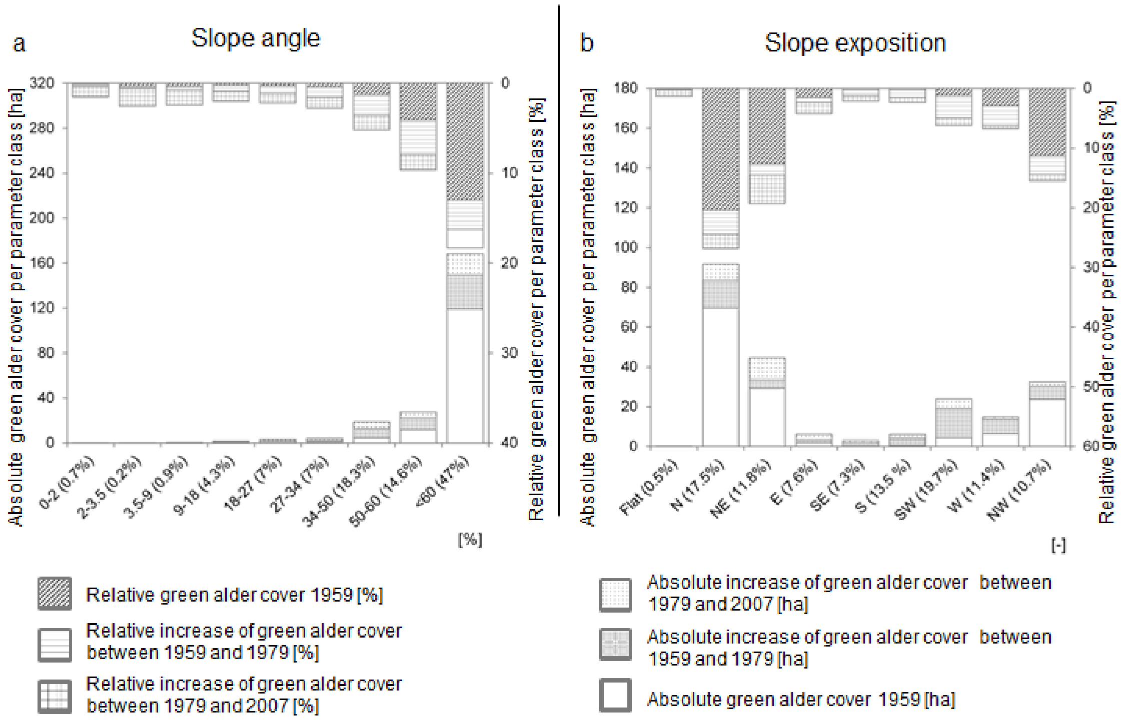


Abbildung 6: Green alder cover and increase per primary relief parameter class: Slope exposition (a) and slope angle (b). Results are shown as absolute area (standing bars) and relative to total vegetated area per defined relief class (hanging bars), calculated for the surface area. The number in brackets indicates the portion of each class to the whole vegetated area in the study are a below 2400 m a.s.l.

The analysis of the secondary relief parameters, namely solar radiation and topographic wetness index (TWI) confirms the encroachment of green alder on areas with drier conditions, mentioned above. The analysis of total solar radiation during the vegetation period illustrated that areas with a solar radiation ˃ 0.5 MWh m^{-2} showed a low coverage of green alder in 1959. On the other hand, more than half of the area within the categories of 0.2-0.3 MWh m^{-2} and 0.3-0.4 MWh m^{-2} was covered with green alder in 1959 (Abbildung 7a). This finding is in accordance to Richard (1969) who described the presence of green alder as limited by the water supply during summer, due to the high evapotranspiration rate of green alder stands. Also the TWI showed slightly higher cover on sites with greater TWI values, representing drainage depressions, in 1959 (Abbildung 7b). However, analyzing the changes in green alder cover between 1959 and 2007, a different pattern can be seen. The greatest increase of 10.3% (21 ha) appeared within the category of 0.4-0.5 MWh m^{-2}. The absolute green alder cover on areas with a solar radiation ˃ 0.5 MWh m^{-2} increased by 67 ha, while green alder cover on areas with solar radiation < 0.4 MWh m^{-2} decreased (-2 ha). The analysis of the TWI on encroached areas revealed a slight trend toward higher proportional increase on areas with a lower TWI, representing drier areas (Abbildung 7b).

Overall, the analysis of green alder encroachment, solar radiation and TWI shows that green alder are spreading on drier areas with higher water evaporation than expected. This finding is in accordance to Eggenberg (2002), as well as Boscutti et al. (2014), who refer to a new ecological association of green alder, colonizing disused pastures on moderate and drier slopes together with alpine rose (*Rhododendrum ferrugineum*). Eggenberg (2002) points out that the presentation of the ecological habitat of green alder, associated with avalanche gullies, moist and steep slopes and characterized by perennial herbs understorey of the *Adenostylion* association has so far been biased. Some authors even suggest a differentiation of the taxon, described as *Alnus brembana* Rota (Landolt, 1993; Lauber and Wagner, 2014), with smaller leaves and minor height as being competitive under drier conditions and spreading on slopes with south-, southwest- and southeast-facing slopes, mostly building transitional populations with *Alnus viridis* (Chaix) DC. [s.str. prov.] (Wettstein, 2001; Senn-Irlet et al., 2012; Huber and Frehner, 2013). In the course of the ground truthing performed for this study, green alder were found both within the *Adenostylion* association (*Achillea macrophylla, Athyrium distentifolium, Adenostyles alliariae, Rumex alpestris, Cicerbita alpina* and *Geranium sylvaticum*) (Anthelme et al., 2007; Bühlmann et al., 2014), but also within the *Rhododendron-Vaccinion* association (*Rhododendron ferrugineum, Vaccinium myrtillis, Juniperus nana* and *Calluna vulgaris* (Eggenberg, 2002; Boscutti et al., 2014). Thus, the suggested differentiation seems reasonable and could explain the observed spreading of green alder on drier more and more moderate slopes.

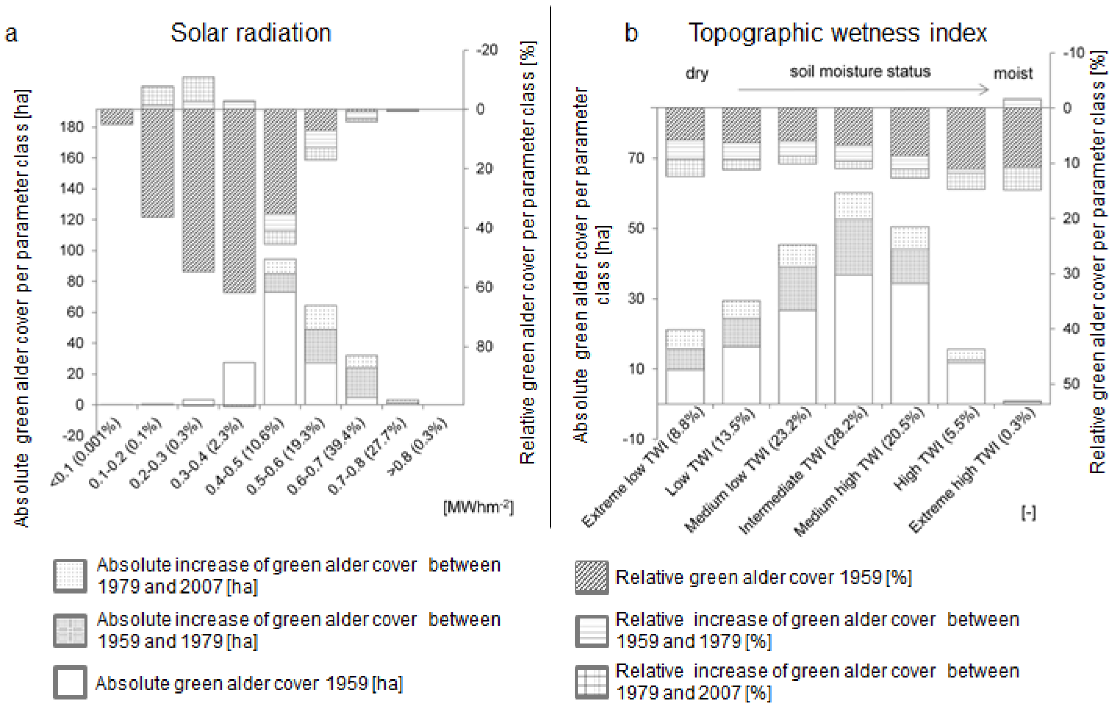


Abbildung 7: Green alder cover and increase per secondary relief parameter class: Solar radiation (a) and topographic wetness index (TWI) (b). Results are shown as absolute area (standing bars) and relative to total vegetated area per defined relief class (hanging bars), calculated for the surface area. The number in brackets indicates the portion of each class to the whole vegetated area in the study area below 2400 m a.s.l.

2.3.3 Geomorphic proxies defining the ecologic niche of green alder

The landform-based analysis of green alder cover and encroachment (Abbildung 8) revealed that in 1959 green alder dominated within channels and debris flow tracks and talus sheets. The green alder encroachment between 1959 and 2007 was highest on debris flow tracks and debris cones. In 2007, landforms which were covered by more than 50% with green alder included channels, debris flow tracks, debris and talus- cones as well as talus sheets (Abbildung 8). Except for the high cover on talus sheets, this finding coincides with the results of Hoersch et al. (2002). Their study in the Western Swiss Alps revealed that the presence of green alder highly correlates with depression zones along channels and debris flow tracks (Hörsch et al., 2002). The authors related their finding to the affinity of green alder to high soil moisture and to the high geomorphic activity of these areas (Hörsch et al., 2002). Geomorphic activity on channels and debris flow tracks, as well as in depositions zones, is considered to be high in the Unteralptal due to the steep slope angle and the high frequency of avalanches and debris flows (Coaz, 1881; Kägi, 1973; Caviezel et al., 2010). In light of the fact that green alder are known to be dominant on disturbed habitats, the result of our analysis is not surprising. However, the area with landforms of high geomorphic activity is restricted and accounts for only 8.4% to the study area. The absolute increase on channels, debris flow tracks, talus- and debris cones amounts to only 16 ha, or 18.4% of the total increase of green alder cover. On the other hand, 76% of the total green alder increase took place on landforms classified as talus sheets, moraine deposits and bedrock, which are considered as less disturbed landforms (Abbildung 8). Thus, the encroachment of green alder on landforms with high geomorphic activity is marginal and encroachment does not exhibit the affinity to disturbed habitats.

The proportion of green alder cover within each landform is significantly higher on the landforms allocated at the orographic left side of the valley. In 1959, 124 ha out of 136 ha of green alder grew on left side allocated areas. The covered area on the orographic right side was almost negligible in 1959, but increased dramatically until 2007 for all landforms (Abbildung 8). A differentiation between relative and absolute increase values provides further findings. The green alder cover on talus sheets on the orographic left side increased by 18 ha, reaching approximately 53% cover in 2007. On the right talus sheet, the cover increased by 8 ha and reached corresponding to an area of just 10.3% within the landform in 2007. Moraine deposits at the right side of the valley show a similar pattern with a high absolute increase of 9.5 ha, but a low proportional cover (2.6%) within the landform in 2007. The low proportional cover within the specific landform and the high absolute increase on talus sheets and moraine deposits on the orographic right side of the valley indicate a high potential for future increase of green alder within the moraine deposits and talus sheet categories.

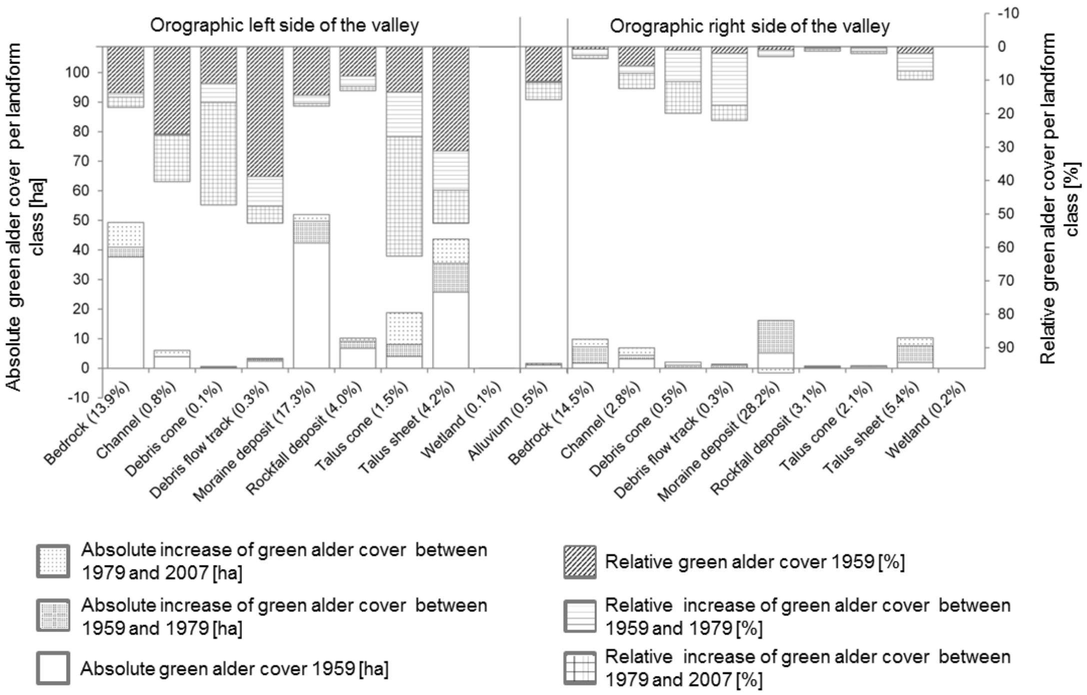


Abbildung 8: Green alder cover and increase per landform class. Results are shown as absolute area (standing bars) and relative to total vegetated area per defined relief class (hanging bars), calculated for the surface area. The number in brackets indicates the portion of each class to the whole vegetated area in the study area below 2400 m a.s.l.

2.3.4 Drivers and ecologic consequences of green alder spreading

The results of the geomorphometric and geomorphic analysis show that green alder encroachment reaches beyond relief parameters values or landforms associated to common ecological requirements described in historic literature. This further than expected spread illustrates that green alder encroachment is not restricted to a narrow ecological habitat and that new limits for present and future green alder cover have to be identified.

The findings of our study indicate that the presence (or absence) of green alder is also controlled by land use intensity. The orographic right side of the valley shows a significantly lower green alder cover for all relief and landform categories over the whole study period. This difference between valley sides might be explained by the more favorable accessibility of pastures. The topographic map of 1915 (Swisstopo, Federal Office for Topography, 1915) already indicates a continuous road on this side of the valley, enabling easy movement of animals, while the left side was separated from the road by the river and only provided discontinuous walking paths. While the walking paths on the left side showed deteriorations over time, the road on the right side has constantly been improved due to the military use and the installation of hydro-electric power plants during the first half of the 20th century (Kägi, 1973). The lower cover on the right side is indicative for a generally more intense grazing due to the better accessibility. However, green alder cover on the easily accessible right side of the valley increased by 34 ha during the observation period. This might indicate the general decrease of land use intensity, where herding small groups of animals became less feasible in the last decades (Wunderli, 2010). An additional factor leading to the observed shrub encroachment on 50-60% slopes is the continuous decline of goat numbers during the 20th century (Caviezel et al., 2010). Over the study period, significant decreases took place during the 1960's and the 1990's (Wunderli, 2010). Further on, the shrub encroachment on 50-60% slopes can be related to the abandonment of steep areas, which have formerly been mown by hand, in the early 1960's (Wunderli, 2010). The marked increase of green alder cover on 50-60% slopes between 1959 and 1979 (11 ha), and the lower, but noticeable increase (5 ha) between 1979 and 2007, would match to the described changes in land use.

Even though land abandonment is known as a driving factor for forest regrowth and shrub encroachment (Bebi and Baur, 2002; Tasser and Tappeiner, 2002; Gehrig-Fasel et al., 2007; Gellrich et al., 2007; M. Gellrich and Zimmermann, 2007; Pellissier et al., 2013), the result of our study highlights the need to redefine the ecological requirements of green alder, and thus the area potentially affected by shrub encroachment. The suggestion of the taxon of *Alnus brembana* Rota (Landolt, 1993; Lauber and Wagner, 2014), described as being competitive under drier conditions and building transitional populations with *Alnus viridis* (Chaix) DC. [s.str. prov.] (Wettstein, 2001; Senn-Irlet et al., 2012; Huber and Frehner, 2013), confirms the knowledge gaps in green alder ecology and geography. It can be speculated that the clearing of the coniferous forest during the 12th and 13th century is a possible explanation for unexpected spreading of green alder nowadays. Throughout agricultural history, mowing and pasturing kept the green alder away from the agricultural land, leading to the misconception that those areas are outside its ecological limits. However, the clearing of the conifers and the centuries of use may now favor the spreading of green alder because an ideal habitat for green alder was created by land use and competing species are missing.

2.4 Conclusion

Historically, the ecological habitat of green alder has been described as naturally restricted to steep, north-facing, moist slopes or depressions with an affinity to high soil moisture and high geomorphic activity (Schröter, 1908; Richard, 1969; Hörsch, 2003). The analysis of the green alder (*Alnus viridis* (Chaix) DC.) encroachment in the Unteralptal, based on relief parameters and landforms associated with this presumed ecological habitat, indicate that the green alder cover is less restricted. Between 1959 and 2007, green alder encroached significant proportions of sunny southwest- to east-facing slopes, as well as on more gentle and drier slopes with lower geomorphic activity. The identification of this "new habitat" boundaries is supported by other studies which identified spread of green alder beyond the conventionally accepted ecological habitat (Wettstein, 2001; Senn-Irlet et al., 2012; Huber and Frehner, 2013; Boscutti et al., 2014).Thus, the habitat spectrum of green alder is much wider than assumed. The spreading of green alder into areas that do not correspond to the previously assumed limited habitat conditions highlights the need to redefine its ecological requirements. The past occurrence or concentration of green alder on steep, north-facing and moist slopes probably coincided with land use patterns that limited its spread to those sites, leading to the misconception that these areas represent an ecological habitat. Considering that land use abandonment and the polarization of land use in alpine areas are likely to continue (Flury et al., 2013), 2013), it can be anticipated that green alder encroachment will be an ongoing process in the Alps. Control measures can be required because green alder can have several concomitant ecological effects. According to the inhibition model presented by Connell and Slatyer (1977), and as shown in different studies (Mallik et al., 1997; Huber and Frehner, 2013), green alder associations are preventing the establishment of other species including coniferous forests. Moreover, green alder does not fulfill the protective function of a conifer forest against erosion (Caviezel et al., 2014). Further ecological effects include the enrichment of the soil with nitrogen (Bühlmann et al., 2016), the associated risk of nitrate leaching into water bodies (Mueller et al., 2016), soil acidification (Caviezel et al., 2014), the increased potential to leach dissolved organic carbon (Hunziker et al., 2017), and the loss of biodiversity (Anthelme et al., 2002; Bühlmann et al., 2014). Based on the presented study, we conclude that the encroachment by the fast spreading green alder plays a major role in land cover change and hence intensively impacts subalpine landscapes and processes.

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Conflict of Interest: The authors declare that they have no conflict of interest.

KAPITEL 3

Shrub encroachment by green alder on subalpine pastures: Changes in mineral soil organic carbon characteristics

Der Blick von der Brücke über die Unteralpreuss talauswärts Richtung Nordwesten zeigt die unterschiedliche Nutzungsintensität an den beiden Talhängen, was zur stärkeren Verbuschung des Osthangs durch die Grünerle führt. Aufgenommen von M. Hunziker am 22. August 2013.

Shrub encroachment by green alder on subalpine pastures: Changes in mineral soil organic carbon characteristics

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Abstract

Shrub encroachment by green alder (*Alnus viridis*) has been an ongoing process in European mountain areas after land abandonment. The invasion of this N_2-fixing and highly productive plant on former subalpine pastures and meadows changes the properties and interactions in the plant-soil system. In the national carbon inventory, it is assumed that green alder woodlands contain the same amount of SOC (~69 t C ha^{-1}) as productive forests due to the lack of data. To explore the rarely studied change of the soil organic carbon (SOC) pools during the shrub establishment, the study used a chronosequence approach by testing pastures in a pre-encroached status and green alder stands with an age of 15, 25, 40 and 90 years, respectively. Besides the estimation of the SOC stock, the soil samples, taken in four different depth layers, were physically fractionated to characterize the quality of the SOC. While pasture grassland contained a median SOC stock (0-30 cm) of 100 t C ha^{-1}, the SOC stocks decreased to 81 t C ha^{-1} for 40 years old shrub stands. The 90 years old green alder bushes showed the significantly highest C stock in the mineral soil with 174 t C ha^{-1}. Green alder encroachment led to an increase of the particulate organic material (POM) in the mineral soil resulting in a high concentration of unprotected carbon. By contrast, a stabilization of the SOC with the mineral soil phase was not visible during green alder establishment. The study concludes that green alder encroachment causes a significant increase of fresh and unprotected carbon in the soil system compared to subalpine pastures, resulting in a significantly higher total SOC stock (+ 74 t C ha^{-1}) after 90 years and a doubling of the potential to leach dissolved organic carbon. At landscape level, the ingrowth and establishment of green alder on abandoned subalpine pastures can therefore influence the terrestrial and aquatic systems. Secondly, the SOC stocks of shrub forests are insufficiently represented in the current National Inventory Report.

Keywords: Green alder, shrub encroachment, SOC stock, SOC quality, soil geochemistry, land abandonment

3.1 Introduction

3.1.1 Land abandonment and forest regrowth in the European Alps with a focus on Switzerland

Over centuries, traditional agricultural cultivation forms generated a unique landscape of high ecological value in the European Alps (Mario Gellrich and Zimmermann, 2007). The resulting landscape is characterized by broad parts of man-made meadows and pastures (Tasser et al., 2011).

During the 20th century political, economic and social changes led to an alteration of the agricultural structures (Flury et al., 2013). Thus, between 1980 and 2000 about 33 % of the farm holdings and 4329 km^2 (-7.6 %) of the agricultural used area have been abandoned within the Alpine arc (Streifeneder et al., 2007; Streifeneder, 2009). In the Swiss Alps, 39% of farms disappeared between 1985 and 2009 (BfS, 2013). As a consequence, the area of sub-/alpine summer pastures in the Swiss Alps decreased by 372 km^2 (-34.5 %) (Schubarth and Weibel, 2013).

Land abandonment is the main driver for forest and shrub regrowth in the Alps which was observed in a wide range of regional studies throughout the whole Alpine arc (Bebi and Baur, 2002; Anthelme et al., 2003; Gellrich et al., 2007; Gehrig-Fasel et al., 2007; Camacho et al., 2008; Cocca et al., 2012; Huber and Frehner, 2013; Pellissier et al., 2013; Caviezel et al., revised). The Swiss National Forest Inventory (NFI) recorded an expansion of the shrub forest of 113 km^2 (+ 21 %) for the Swiss Alps (NFI categories "Alps" and "Southern Alps") between the observation periods 1983/85 and 2009/13 (WSL, 2015a, 2015b). This increase was proportionally higher than the regrowth of mature forests (+18 %) (WSL, 2015a, 2015b). Currently, shrubs cover an area of about 644 km^2 in the Swiss Alps (WSL, 2015b). Besides dwarf mountain pine (*Pinus montana* subsp. *prostrata* Tubeuf), hazel (*Corylus avellana* L.) and various willow species, 70% of the shrub forest area consists of green alder (*Alnus viridis (Chaix) DC* = *Alnus alnobetula (Ehrh.) K. Koch*).

A. viridis is an early successional and fast expanding species (Cioldi et al., 2010). According to Schröter (1908) and Richard (1968), green alder is naturally restricted to steep, north facing subalpine well drained slopes and debris flow tracks. However, recent studies in the eastern and central parts of Switzerland have shown, that the ecologic niche of green alder is much wider than assumed and that the abundance of green alder is mostly controlled by land use intensity (Huber and Frehner, 2013; Caviezel et al., revised). Consequently, green alder bushes have the potential to spread on abandoned subalpine pastures and influence the vegetation and soil system of larger areas than presumed so far (Caviezel et al., revised).

3.1.2 Impact of green alder encroachment on carbon stocks, carbon quality and soil properties

Available data for common carbon pools of the shrub forest or even for the green alder is sparse. Thus, the National Inventory Report (NIR) assesses equally the carbon stock values of productive forests and those of shrub forests (69 t C ha^{-1}), except for the living above-ground biomass pool (FOEN, 2015). For the green alder species, Bühlmann et al. (2016) recently reported 26 t C ha^{-1} in the above-ground biomass, 8 t C ha^{-1} in the below-ground biomass, 1 t C ha^{-1} in the fresh litter and 62 t C ha^{-1} in the mineral SOC stocks for bushes growing at 1650 m asl. The annual biomass productivity of green alder communities was estimated to be 6 t ha^{-1}, which is higher compared to other subalpine vegetation species such as *Picea abies* and *Betula pendula* or grassland (Wiedmer and Senn-Irlet, 2006; FOEN, 2014, 2015; Bühlmann et al., 2016). The different biomass production of *A.viridis* communities consequently results in a change of the quantity and quality of litter material as well as the carbon contained in the litter when pasture land turns into green alder shrubland (Jackson et al., 2007). Furthermore, under the newly established green alder bushes, the decomposition of the litter and its residues is hampered by i) the cool and moist conditions due to the shade of green alder canopies (Bühlmann et al., 2014, 2016) and ii) the increase of the C/N ratio due to the higher proportion of woody material, which is unfavorable to the detritus consumers (Jackson et al., 2007; Bühlmann et al., 2016) (Abbildung 9).

The ingrowth of green alder on abandoned grassland areas strongly influences the soil properties and its quality, hence the ecosystem services, from a more eco-pedological perspective (Bühlmann et al., 2014; Caviezel et al., 2014; Hiltbrunner et al., 2014; Bühlmann et al., 2016) (Abbildung 9). Due to the symbiosis between *A. viridis* and the actinomycete *Frankia alni*, the soil experiences an enrichment of nitrogen (Bühlmann et al., 2014). Thus, the soil processes under green alder stands are hardly comparable to those under forest or other vegetation types. The soil nitrogen enrichment by N_2 fixation can hamper the succession of coniferous species and the green alder vegetation type can build a long persisting climax vegetation due to unfavorable soil nutrient composition (Anthelme et al., 2002; Huber and Frehner, 2013; Hiltbrunner et al., 2014). The increased N pool also creates the potential for nitrate leaching to water bodies (Bühlmann et al., 2016; Mueller et al., 2016).

Another possible change in soil chemistry induced by green alder is the substantial decrease of the pH value and thus, the base saturation of the encroached soil as a consequence of the nitrogen enrichment and the proton production during the nitrification process under green alder stands (Podrazsky and Ulbrichova, 2003; Caviezel et al., 2014; Bühlmann et al., 2016). Green alder encroachment also leads to a decline in root density while soil porosity and SOC concentration increase (Caviezel et al., 2014) (Abbildung 9).

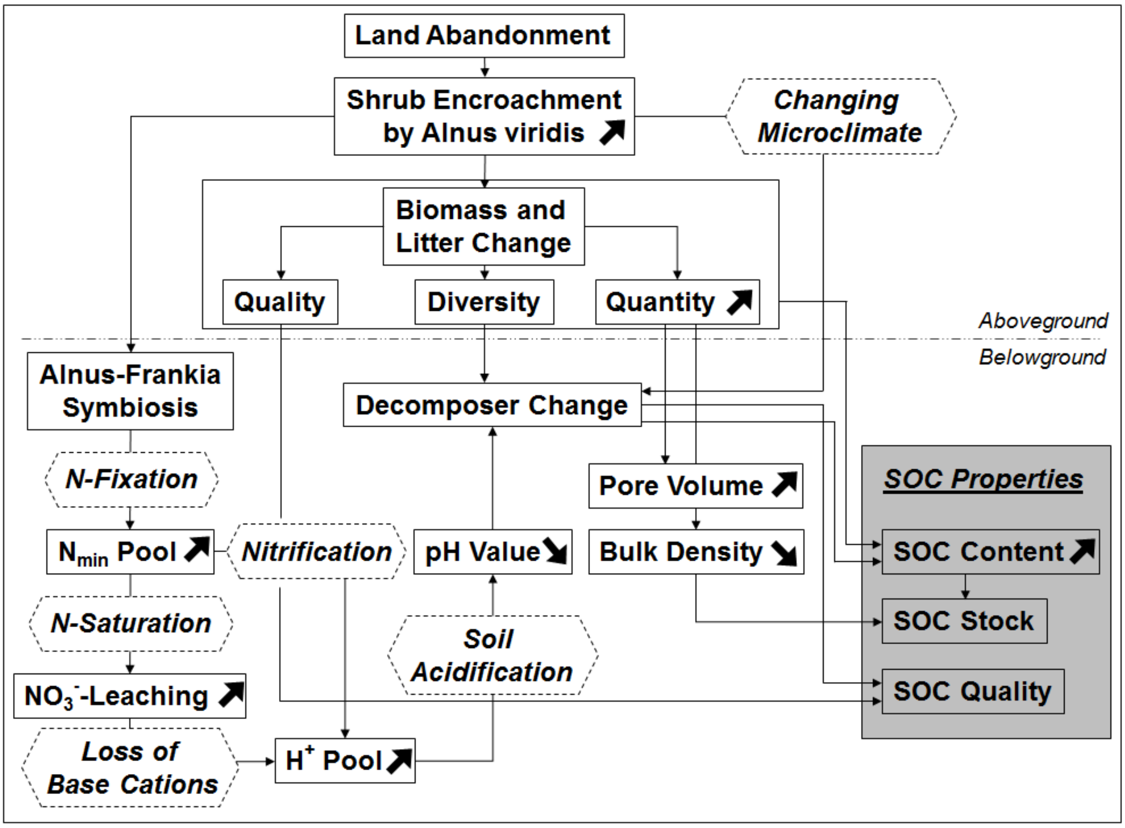

Abbildung 9: The hypothetical model shows how the SOC properties are influenced by the *A. viridis* encroachment hence, other soil properties and processes (italic). Based on the findings of Caviezel et al. (2014), Bühlmann et al. (2016) and Caviezel et al. (revised), the arrows show the direction of measured changes induced by the green alder encroachment.

3.1.3 Aims and objectives of the study

The on-going green alder encroachment potentially leads to an unprecedented change in the soil conditions that is difficult to compare to other types of land cover and change. Further, the ingrowth can affect the sequestration and hence the residence time of the soil organic matter components in different functional pools which are characterized by distinctive stabilization mechanisms (von Lützow et al., 2007; Schmidt et al., 2011). Due to the strong linkage between the biomass as well as litter characteristics and the SOC, there is need to fill these knowledge gaps regarding the change of the SOC quantity and quality during the establishment of *A. viridis* shrubland. Additionally, other soil properties might be influenced by the establishment of high-productive green alder shrubland and hence the expected SOC changes.

The present study therefore aims to answer the following research questions (i) What happens to the SOC stock during the encroachment process by green alder? (ii) Is the SOC stock of green alder stands comparable with the assumed SOC stock of 69 t C ha^{-1} in the NIR? (iii) How does the SOC quality change during the encroachment? (iv) And how can relationships between SOC properties and other soil properties (Abbildung 9) change and thereby influence landscape-level processes? To answer these questions, the SOC quantity and quality of differently aged green alder stands were compared with the pasture sites by applying a chronosequence approach and using physical SOC fractionation techniques. Furthermore, the relationships between the SOC properties and other measured soil parameters, which are relevant in this context, were tested.

3.2 Material and methods

3.2.1 Study area

The Unteralptal (35 km²) is a side valley of the main Urserntal in the central Swiss Alps. Due to land-use change, green alder encroachment has been an ongoing process in the Unteralptal during the last decades (Caviezel et al., 2014, revised). The bottom of the valley is located in Andermatt at 1442 m asl. The elevation range is 1458 m (Ambühl et al., 2008). The mean annual air temperature is 3.4 °C and the total amount of precipitation is 1422 mm per year at the MeteoSwiss weather station in Andermatt. The mostly NW-facing and U-shaped valley is characterized by a rugged terrain due to glacial, fluvial and gravitational processes (Ambühl et al., 2008; Caviezel et al., revised). Based on the FAO World Reference Base for soil resources (Food and Agriculture Organization of the United Nations, 2006), the following soil types are found in the valley: Leptosols on middle slope positions, Padzdocambisols and Cambisols on lower slope positions and Fluvisols and Gleysols at the valley bottom. Traditional summer pasturing has influenced the vegetation composition since the 14th century. In order to obtain summer pasture areas, the coniferous forest has been mostly cleared in the 13th century (Kägi, 1973; Küttel, M., 1990). According to travel reports in the 18th century, only few and small forests were observed in the area (Scheuchzer, 1746; Kasthofer, 1822). A pre-examined survey of the land cover change showed that grassland and dwarf shrub areas decreased by 4.6 % and 12.9 %, respectively, between the years 1959 and 2007. On the other hand, the areas with green alder cover increased by 63.3 %. Thus, in 2007 the vegetated land cover (< 2400 m a.s.l.) was composed of (sub-/alpine) pastures and grassland (1710 ha), dwarf shrub associations (22 ha) and green alder associations (223 ha) (Caviezel et al., revised).

3.2.2 Shrub encroachment detection, selection of sampling sites and its floristic classification

The chronosequence approach was used for the soil sampling scheme. Chronosequences are appropriate to study plant-soil feedbacks and temporal changes in the soil when i) there is an evidence of site history and ii) that sites of different ages are following the same trajectory (Walker et al., 2010). In this case, the management of the actually grazed areas in the valley has not changed significantly for the last 700 years (Kägi, 1973; Caviezel and Kuhn, 2012), therefore this space for time analogy is acceptable. Based on the space for time approach, present grazing areas were selected to describe the status (Control) before bush encroachment begins. To ensure the above-mentioned criteria for using a chronosequence approach, different independent dating techniques were employed to define the age classes of the chronosequence. Firstly, we used aerial photographs dated from 1979, 1993 and 2007 and a topographic map released in 1926 to identify areas of different encroachment ages. Secondly, stem disks of the thickest stems were taken at each sampling site during field work. The used dendrochronology technique enabled an even more precise age determination than aerial photographs, at least for the oldest surviving plants. The mean ages of the sampled stands were 15, 25, 40 and 90 years, respectively. Thus, the differently aged encroachment stages of the chronosequence were defined as Control, GA15, GA25, GA40 and GA90 (Abbildung 10). More detailed information concerning the site selection process is found in Caviezel et al. (2014).

The vegetation compositions at the different encroachment stages were classified on the association level according to Mertz (2008) and Delarze et al. (2015). The pasture sites (Control) were mostly allocated to the Poion alpinae association (e.g. Phleum *alpinum, Festuca rubra* and *Ranunculus montanus*). Green alder bushes of the GA15 stage were mostly growing within the *Adenostylion* association (*Achillea macrophylla, Athyrium distentifolium, Adenostyles alliariae, Rumex alpestris, Cicerbita alpina* and *Geranium sylvaticum*) but also within *Poion alpinae* association and the *Rhododendron-Vaccinion* association (*Rhododendron ferrugineum, Vaccinium myrtillus, Juniperus nana* and *Calluna vulgaris*). At the stage of GA25, green alder bushes were predominantly sampled in *Adenostylion* association, but sporadic in the *Poion alpinae* association. Under the closed canopies of green alder formations (*Alnenion viridis* association) at GA40 and GA90, the understory vegetation was represented by species of the *Adenostylion* association and partly by *Rhododendron ferrugineum* (only found at GA40).

3.2.3 Soil sampling

The field work was carried out in July and August 2011 except for the sites of GA90 where soil samples were taken in October 2011 and August 2013. The number of chronsequences and hence the number of sites per age class (replicates) was N = 9 (Abbildung 10). At each test site, five soil pits were placed. At the encroached locations, the soil pits were placed within one half of the crown diameter of the selected green alder bush (Abbildung 10). After removing the organic layer, the top 30 cm of the mineral soil was sampled which represents the common depth for SOC stock inventories (Aalde et al., 2006b; FOEN, 2015). The applied sampling depth contains most of the below-ground living root biomass at pasture and green alder areas, respectively, representing the below-ground source for SOC (Caviezel et al., 2014). Due to the mountainous environment, rock fragments lead further to the development of shallow soil types (see below). To ensure the vertical resolution, the analyzed soil profile was split in four depth intervals of 0-5, 5-10, 10-20 and 20-30 cm, respectively. For the estimation of the soil organic carbon stock, the soil samples were taken by a metal cylindric core (Eijkelkamp Soil & Water, Giesbeek) of 100 cm³ volume and 5 cm length. Due to the high amount of coarse soil material and rock fragments within certain sampling depths, it was not always possible to sample the soil with the corer or even take a soil sample. Thus, the dataset consists of 863 soil samples with just 506 samples taken by coring. The remaining 357 samples were obtained without recording the volume. For the carbon stock calculation, only 506 complete samples were used.

The study is focused on the SOC change along the encroachment time and less on the spatial heterogeneity at plant scale. Thus and in order to make the dataset more robust, we mixed an equal amount of soil from each replicate sample (N=5), taken in a certain soil depth at a certain test site, to a composite sample before performing the SOC fractionation procedure. Based on this, 180 mixed samples were fractionated. Thus, each depth class per age class is represented by 9 samples.

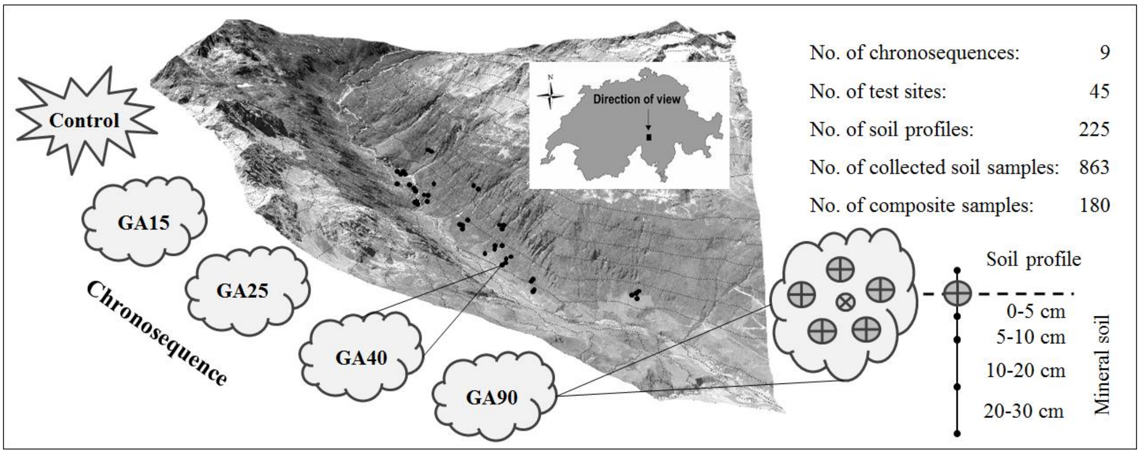

Abbildung 10: The Unteralptal as the study area is located south of Andermatt (Switzerland). A cluster of 4 black dots (test sites) indicates the control site and the age classes GA15 to GA40 of one chronosequence. The higher elevated dots represent the test sites of GA90. At each test site, five replicate samples were taken in the given four depth layers.

3.2.4 Laboratory analysis on the bulk samples and soil fractionation

Soil preparation

In the laboratory, all 863 soil samples were dried at 40 °C until a constant weight was reached. The weight, volume and bulk density (BD) of the fine earth (< 2mm) were determined by dry sieving and water displacement of the coarse material (> 2mm).

Soil fractionation

The responses of SOC stability to land-use changes can be investigated through physical fractionation methods, which allow to study the organization of OM within the soil matrix (Cambardella and Elliott, 1992; Six et al., 1998; Poeplau and Don, 2013). We therefore slightly modified the separation procedure presented by Zimmermann et al. (2007b) and used the disaggregation, the particle size separation and the density fractionation to separate the SOC into the given functional groups (von Lützow et al., 2007). The fractionation procedure was performed with the material of the fine earth (< 2mm) of the 180 composite samples. Initially, the samples were dispersed by an ultrasound treatment (22 J ml^{-1}) in 150 ml deionized water to retrieve only primary organo-mineral complexes (Christensen, 2001). Afterwards, the samples were wet sieved at 63 µm to separate the stable sand-sized aggregates and the un-protected particulate organic matter (POM) from the silt and clay particles (S+C). Subsequently, the POM material was separated from the denser organic material in the mineral-associated sand and aggregate fraction (HF) by density fractionation (2.0 g cm^{-3}, SPT from Sometu) on the soil material (> 63 µm). After separation, both fractions were washed with deionized water until the electrical conductivity of the rinse water reached < 50 µS (Wagai et al., 2008). The material smaller than 63 µm represents the SOC pool of the silt and clay fraction (S+C). Further, after settling time, a sample of the suspension (< 63 µm) was taken, filtered with a 0.45 microns filter and analyzed for its dissolved organic carbon content (DOC). The value of the DOC concentration was used as an indication for the ability of the studied soils to leach dissolved organic carbon. Compared to Zimmermann et al. (2007b), the present study used a SPT density of 2.0 g cm^{-3} and did not conducted the oxidation with sodium hypochlorite (NaOCl).

Measurement of the carbon content

All samples of the bulk soil (< 2mm) and the soil fractions were ball-milled and then analyzed for organic carbon content by the dry combustion technique (Leco CN 628 Elemental Determinator). The dissolved organic carbon was measured by the combustion analytic oxidation method (TOC-5000A, Shimadzu).

Calculations and statistics

Besides the analysis of the parameters of the SOC properties, further soil properties such as bulk density, pore volume, pH value, shear resistance and penetration resistance were recorded. Sampling and analysis methods concerning these properties are explained in detail in Caviezel et al. (2014).

The amount of soil organic carbon that is stored in a given soil profile is defined as the SOC stock and is expressed in tons per hectare. According to Ellert et al. (2008) and Rodeghiero et al. (2009), the SOC stock (SOC_{stock}; t C ha^{-1}) is a function of the soil's carbon content (SOC %; [-]), the bulk density (BD; g cm^{-3}) of the fine earth (< 2mm) and the investigated soil depth (l; cm). The conversion factor between the units is 100.

$$SOC_{stock} = SOC\% \times BD \times l \times 100 \qquad \text{(Eq. 1)}$$

The statistical analysis was performed in SPSS (22.0) statistical package (IBM Corp.). Besides the descriptive statistics, we graphically and numerically tested the data of the different age and depth classes for normality. Due to the skewed distributions, the descriptive statistics showing the median values and the non-parametric Mann-Whitney rank sum test was further used to detect significant (significance level: $p = 0.05$) changes in the SOC stocks and the SOC fractions between the different age classes. To test the relationships between the SOC properties and the other measured soil parameters (Caviezel et al., 2014), the Spearman Rho test was employed as a bivariate correlation test with a significance level of $p = 0.05$. By applying the Spearman Rho test, the age classes were treated separately to detect possible changes of relationships during shrub encroachment.

3.3 Results

3.3.1 The SOC stocks during the establishment of green alder forest

Pasture areas contained a median SOC stock (0-30cm) of 100 t C ha^{-1} (Abbildung 11). During green alder encroachment, the SOC stock tended to decrease within the first 40 years after the vegetation shift, while the decline mostly occurred in the top 20 cm. However, the changes were not significant (Mann-Whitney rank sum test; $p < 0.05$). After 90 years of green alder cover, the SOC stock was higher compared to those of the pasture sites and even the younger green alder stands. These differences were significantly higher in the top three sampling layers (Abbildung 11). Compared to the SOC stock of the National Inventory Report (69 t C ha^{-1}), the SOC stocks of the tested and differently aged green alder stands were considerable higher (> 12 t C ha^{-1}).

Annual SOC stock changes varied accordingly, within 40 years after the vegetation shift, former grassland soils annually lost between 0.03 and 0.60 t C ha^{-1}. After this, a gain of SOC was observed by approximately 1.86 t C ha^{-1} until the stand reached the age of 90 years. Overall, the encroachment of subalpine pastures to a 90 years old green alder shrubland led to an average annual SOC sequestration rate of 0.86 t C ha^{-1} yr^{-1}.

3.3.2 The SOC concentration in the studied fractions

The SOC pools also showed distinct patterns of change with an increasing duration of shrub cover. In each depth layer, the analysis revealed a significant increase of the SOC concentration in the particulate organic matter (POM) fraction between the pasture sites and the 90 years old green alder stands (GA90). The younger green alder sites mostly differed significantly from GA90 and were statistically comparable with the control sites (Abbildung 12; A, E, I, M).

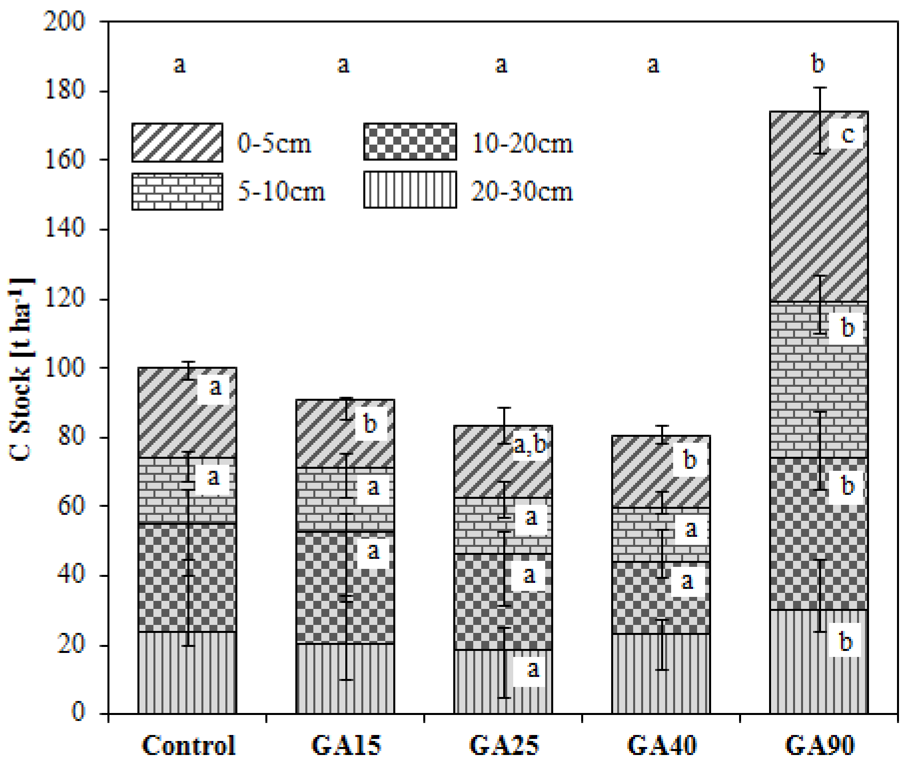

Abbildung 11: The median soil organic carbon stocks [t C ha^{-1}] in the mineral soil along the studied chronosequence (N=9). The range of the error bars is showing the data range between first and third quartil. The different shadings indicate the four sampling depths (0-5cm: diagonal lines; 5-10cm: rectangular squares; 10-20cm: b,w squares; 20-30cm: vertical lines). Within a sampling depth, significant differences (Mann-Whitney U Test, $p < 0.05$) between the age classes are designated by different letters. Further, significant differences (Mann-Whitney U Test, $p < 0.05$) between the total studied soil depth (0-30cm) are shown above the stacked columns by different letters.

The change of the SOC concentration of the mineral-associated sand and aggregate fraction (HF) did not show a trend along the chronosequence but was depth-dependent. In the two upper most sampling depths, a decrease of the SOC concentration was observed and the depletion of the SOC concentration was significant between the pasture sites and GA90 in the top 5cm. On the other hand, an increase of the SOC concentration in the HF fraction was measured in the two deeper sampling depths and the increase was significant in 20-30 cm after 90 years (Abbildung 12; B, F, J, N).

The SOC concentration in the S+C fraction generally showed an increase for all studied sampling depths. Thereby, the GA90 sites significantly differed from the pasture sites in the two upper most sampling depths. However, the change was not uniformly related to the age of the shrubs. Because the SOC concentration showed to decrease during the first 15 years of shrub growth following by a continuous increase between 15 and 90 years of the stand. Thus, the SOC concentration at GA15 was significantly lower than the concentration at GA90 in all depth classes (Abbildung 12; C, G, K, O).

The analysis of the DOC concentrations revealed a continuous increase of the concentration in all studied depth layers. The increase was significant after 40 (5-10cm) and 90 years (0-5 and 10-20cm), respectively, compared to the pasture sites (Abbildung 12; D, H, L, P).

The study also cumulated the SOC concentrations of the analyzed fractions for the entire studied soil profile (0-30 cm). The POM fraction exhibited by far the highest SOC concentrations (Abbildung 13). The ingrowth and establishment of green alder shrubs on former pasture land caused an increase of the SOC concentrations (0-30 cm) in the POM, S+C and DOC fractions. At GA90, the concentrations in the POM and DOC fractions were significantly higher than those at the younger shrub stands and the sites on pasture land. The SOC concentration in the HF did not indicate a clear trend but tended to decrease over time (Abbildung 13; A-D).

3.3.3 The SOC stock stored in the SOC fractions

After the vegetation shift from grass to shrub, the SOC stocks of the analyzed fractions showed no significant decreases (Tabelle 4). Further, the POM fraction accounted for > 50 % of the total C stock (0-30 cm) for each studied age class (Tabelle 4). After 90 years of green alder vegetation, the POM fraction stored the highest amount of the stock, circa 115 t C ha^{-1}. In addition, the S+C fraction at GA90 recorded a higher SOC stock compared to the pasture sites. However, the C stock of the HF fraction showed a significant reduction during the establishment of shrubland and remained after 90 years of shrub growth on a lower stock level compared to the control sites. The values of the DOC stock showed a significant increase and almost a doubling between the grassland and the oldest green alder stands.

Tabelle 4: The median SOC stocks [t C ha^{-1}] for 0-30 cm explained by the analyzed SOC fractions. The parentheses are showing the value range between the first and the third quartile. Within each fraction, different letters designate significant differences based on the Mann-Whitney rank sum test for non-parametric data ($p < 0.05$).

Age Class	SOC stock [t C ha^{-1}]			
	POM	HF	S+C	DOC
Control	52.8 (2.7) ^{a}	11.5 (2.7) ^{a}	34.1 (7.1)	1.6 (0.1) ^{a}
GA15	37.0 (69.7) ^{a}	8.2 (6.3) ^{a,b}	25.1 (23.3)	1.5 (0.9) ^{a}
GA25	46.8 (38.2) ^{a}	6.5 (4.0) ^{b}	25.2 (18.8)	1.7 (0.9) ^{a}
GA40	43.5 (34.2) ^{a}	6.1 (3.4) ^{a,b}	35.3 (18.8)	1.8 (1.0) ^{a,b}
GA90	115.2 (67.2) ^{b}	7.0 (7.6) ^{a,b}	49.5 (29.1)	2.7 (3.1) ^{b}

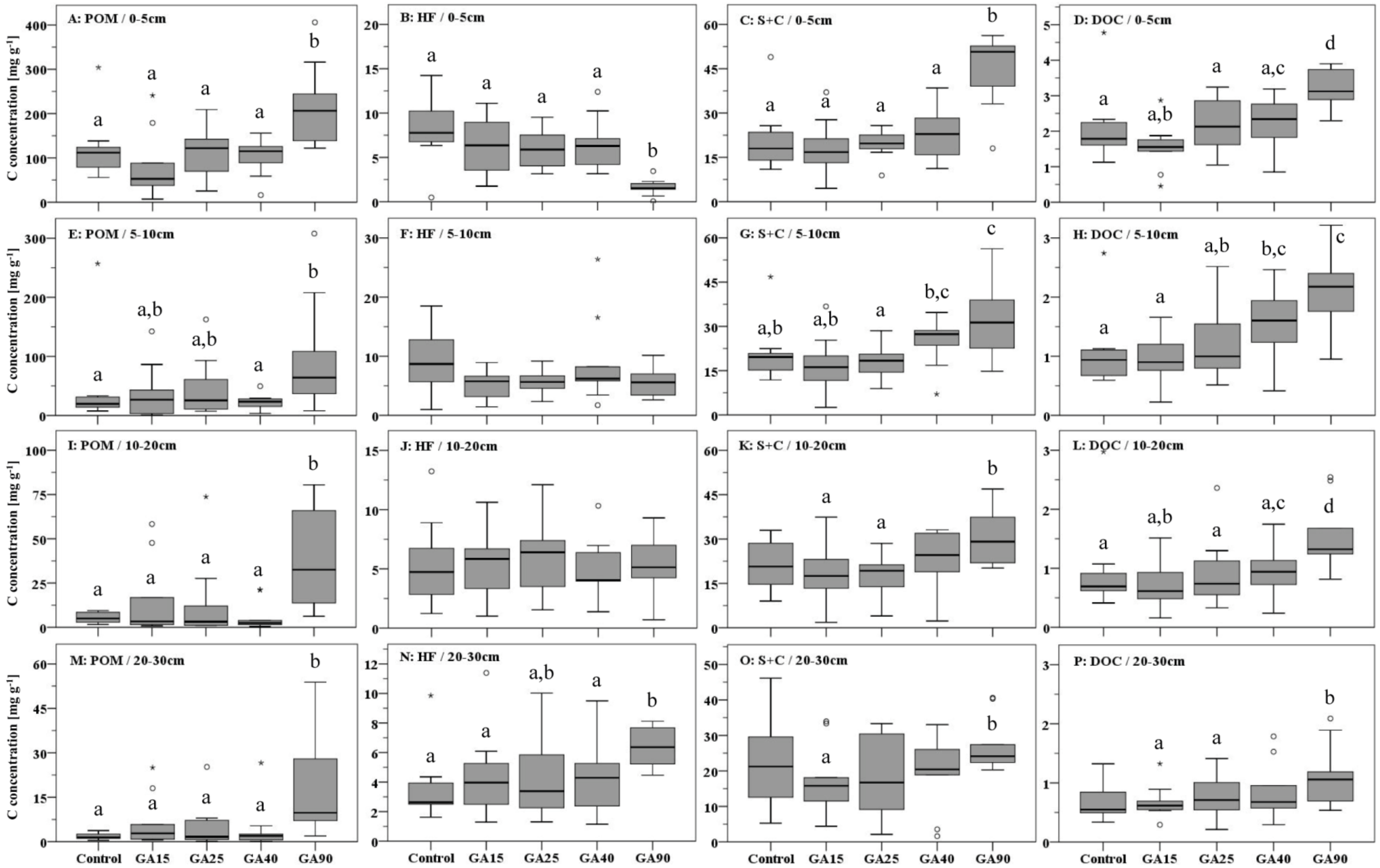

Abbildung 12: SOC concentration [mg g^{-1}] of the fraction POM (A, E, I, M), HF (B, F, I, N), S+C (C, G, K, O) and DOC (D, H, L, P) divided into the sampled soil depths (0-5, 5-10, 10-20 and 20-30 cm). Notice that the scale of the Y-axis is variable. Different letters within a sampling depth and fraction designate significant differences between the age classes (Mann-Whitney U Test, $p < 0.05$).

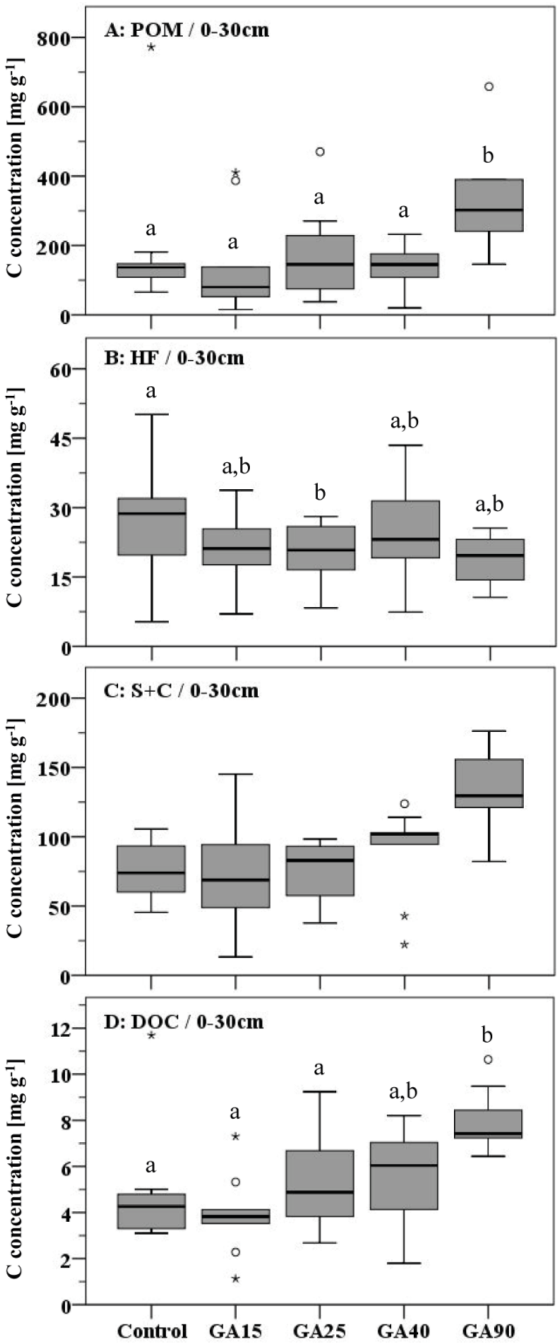

Abbildung 13: Cumulated carbon concentrations [mg g^{-1}] (0-30cm) within the analyzed SOC fractions along the age classes of the applied chronosequence. Notice that the scale of the Y-axis is variable. Different letters per fraction designate significant differences between the age classes (Mann-Whitney U Test, $p < 0.05$).

3.3.4 Relationships between the SOC properties and other soil properties

In the following section, the significant correlations are indicated by a star (*). The SOC stocks correlated positively at a moderate level with the carbon concentration (SOC %) of the bulk soil (< 2 mm) ($R > 0.34$; $p > 0.05$ at GA40). The soil bulk density weakly to moderately corresponded to the SOC stock (R: -0.54 – -0.24; $p > 0.05$ at GA15 and GA40). For all age classes, the correlation analysis between the mass of the POM material and the soil bulk density indicated very strong negative and significant relationships ($R > -0.78$, $p < 0.05$). During the shrub encroachment, the increasing amount of fresh and less decomposed organic matter given by the POM mass indicated that a negative relationship exists between the organic material and the soil stability. This is showed by the relationships between POM mass and the torsional shear strength (R: 0.11, -0.62*, -0.80*, -0.60*, -0.31) as well as the penetration resistance (R: -0.05, -0.61*, -0.69*, -0.66*, -0.65*).

Further, the POM-C concentration very strongly and significantly correlated with the DOC concentration ($R > 0.82$, $p < 0.05$) for grassland and green alder sites. However, HF-C (R: 0.19, 0.48*, 0.40*, 0.67*, -0.63*) and S+C-C (R: 0.46*, 0.57*, 0.61*, 0.53*, 0.54*) showed also significant correlations with the DOC concentration after the vegetation change. As expected, the study found no correlation between the pH value and the DOC concentration at the grassland sites (R: -0.02). However, the gradient of the negative correlation increased with shrub age and ended with a very strong and significant relationship at GA90 (R: -0.58*, -0.47*, -0.78*, -0.86*). Independent of vegetation cover or shrub age, the DOC concentration very strongly and significantly corresponded with soil pore volume ($R > 0.86$, $p < 0.05$) and soil bulk density ($R > -0.74$, $p < 0.05$).

3.4 Discussion

3.4.1 Land-use change by land abandonment can influence the total SOC stock

The SOC stocks (0-30 cm) of the tested subalpine pastures and the differently aged green alder stands varied. The pasture grassland showed a median SOC stock of 100 t C ha^{-1}. After the vegetation shift to *A. viridis* shrubland, the stocks decreased to 81 t C ha^{-1} in 40 years old shrub stands. After that, the significant increase resulted in a SOC stock of 174 t C ha^{-1} in 90 years old green alder bushes (Abbildung 11). Similar SOC stocks (0-30 cm) of subalpine pasture land (81 t C ha^{-1}) and different old green alder stands (62-68 t C ha^{-1}) in the Alps of central Switzerland were recently published by Bühlmann et al. (2016) (Tabelle 5). Compared to the SOC stock (0-30 cm) of 69 t C ha^{-1} for bush forests and 75 t C ha^{-1} for grassland (> 1200 m asl) in the National Inventory Report (FOEN, 2015), the SOC stocks of the tested sites in the present study were considerably higher (> 12 t C ha^{-1}). Recently, Gosheva at al. (2017) studied the SOC stocks of different aged broadleaf and coniferous forests and concluded that forest expansion on former grassland is not associated with an increased C sequestration in the mineral soil, instead a decrease of the SOC stock was reported.

For mountain soils, Sjögersten et al. (2011) reported a median SOC stock of circa 100 t C ha^{-1} which is comparable with the SOC stock for subalpine pasture land of the present study. As the authors showed in their review (Sjögersten et al., 2011), the SOC stocks of mountain soils are highly variable due to elevation and temperature on regional scale (Leifeld et al., 2005; Djukic et al., 2010) and topography, lithology and vegetation type on local scale (Leifeld et al., 2005; Egli et al., 2009; Djukic et al., 2010; Caviezel et al., 2014b; Hoffmann et al., 2014b; Bühlmann et al., 2014b).

Looking at the SOC stock of the mineral soil during the development of the secondary vegetation after land abandonment, available studies do not show clear results (Vesterdal et al., 2011). Risch et al. (2008) found no significant effect of secondary succession by coniferous species on mineral SOC at high elevation (Tabelle 5). Similar findings were reported by a study on shrub encroachment in the Iberian peninsula (Montané et al., 2007). Meyer et al. (2012) concluded that converting mountain grassland into shrubland in the Eastern European Alps does not seem to be a C sink in the soil. However, Alberti et al. (2008) reported a drastic decline of the SOC stock within 75 years after the beginning of the secondary succession by mixed ash (*Fraxinus excelsior* L.) and sycamore (*Acer pseudoplatanus* L.) in the Eastern Prealps of Italy. In the Italian Alps, Guidi et al. (2014b) also found a clear decline of the SOC stock after the abandonment of managed grasslands followed by the afforestation by Norway spruce (*Picea abis* (L.) Karst.) (Tabelle 5). Further, field studies of Thuille and Schulze (2006) and Hiltbrunner et al. (2013) observed first a decline, followed by an increase of the mineral SOC stock after the conversion of grassland into coniferous forests. This pattern is supported by the model equation of Poeplau et al. (2011) based on data of 16 references.

According to Guo and Gifford (2002) and Vesterdal et al. (2013), temporal patterns of the SOC stock after the vegetation shift are the consequences of the changed ratio between biomass and litter input into the soil and the output of carbon by oxidation and outwash. Due to the change of the litter quality (Montané et al., 2010), the vegetation shift from grassland to wooded land commonly causes a vertical change of the SOC pool in the soil profile (Vesterdal et al., 2013). Thus, it can lead to a decrease of the SOC stock in the mineral soil, which can be compensated by a new developed or more distinct organic layer as carbon pool (Guo and Gifford, 2002; Thuille and Schulze, 2006; Hiltbrunner et al., 2013; Guidi et al., 2014b; Gosheva et al., 2017). In the present study, we only studied the mineral soil phases. Contrary to other studies (Hiltbrunner et al., 2013; Poeplau and Don, 2013; Guidi et al., 2014a; Gabarrón-Galeote et al., 2015; Bühlmann et al., 2016), we were, however, able to show an increased SOC stock in the mineral soil after 90 years of green alder growth (Abbildung 11). Our explanations for the significantly higher SOC stock at the mature A. viridis stands are: i) the higher C input by the high above- and below-ground biomass productivity of the green alder ecosystem (Wiedmer and Senn-Irlet, 2006), ii) the nitrogen-fixation leads to an increase of the below-ground phytomass material which contains a lower C:N ratio (Bühlmann et al., 2016) that is less favorable as energy source for decomposers and iii) the vertical transportation of organic material into the mineral phases due to the higher soil porosity (Caviezel et al., 2014) (see chapter 3.4.3).Therefore, our results report an annual loss between 0.03 and 0.60 t C ha^{-1} (until GA40) followed by a gain of 1.86 t C ha^{-1} (Abbildung 11). Hence, high productive green alder systems can reach the pre-shrub status of the SOC stock earlier than 80 years (Abbildung 11), which was reported as a minimum time span for the recovery of the initial SOC stock level during forest establishment (Thuille and Schulze, 2006).

Tabelle 5: Literature review of studies focusing on the change of the SOC stock [t C ha^{-1}] for given mineral soil depth layers in relation to land-use changes in the European Alps and the Pyrenees.

Habitat	Country	Elevation [m asl]	Depth [cm]	SOC stock [t C ha^{-1}]	Reference
Pasture	CH	1580-1750	0-30	100	Present study
Green alder stands (15yrs)	CH	1580-1740	0-30	91	Present study
Green alder stands (25 yrs)	CH	1610-1730	0-30	84	Present study
Green alder stands (40 yrs)	CH	1620-1750	0-30	81	Present study
Green alder stands (90 yrs)	CH	1670-1890	0-30	174	Present study
Montane pasture	CH	1650	0-30	79	Bühlmann et al. 2016
Montane pasture	CH	1950	0-30	81	Bühlmann et al. 2016
Green alder stands (25-100 yrs)	CH	1650	0-30	62	Bühlmann et al. 2016
Green alder stands (50 yrs)	CH	1950	0-30	68	Bühlmann et al. 2016
Mesic subalpine grassland	E	1650-2100	0-30	166	Montané et al., 2007b
Subalpine broom shrub areas	E	1650-2100	0-30	178	Montané et al., 2007b
Subalpine juniper shrub areas	E	1650-2100	0-30	177	Montané et al., 2007b
Meadow	I	600-1200	0-30	86	Guidi et al., 2014b
Meadow w. shrubs and Norway spruce (10 yrs)	I	600-1200	0-30	72	Guidi et al., 2014b
Norway spruce forest (35 yrs)	I	600-1200	0-30	51	Guidi et al., 2014b
European beech and Norway spruce forest (>150 yrs)	I	600-1200	0-30	41	Guidi et al., 2014b
Pasture	CH	1450-1700	0-30	133	Hiltbrunner et al., 2013
Afforested spruce forest (25 yrs)	CH	1700	0-30	134	Hiltbrunner et al., 2013
Afforested spruce forest (30 yrs)	CH	1520-1610	0-30	139	Hiltbrunner et al., 2013
Afforested spruce forest (40 yrs)	CH	1520	0-30	101	Hiltbrunner et al., 2013
Afforested spruce forest (45 yrs)	CH	1450	0-30	102	Hiltbrunner et al., 2013
Spruce forest (>120 yrs)	CH	1450	0-30	130	Hiltbrunner et al., 2013

Fortsetzung

Habitat	Country	Elevation [m asl]	Depth [cm]	SOC stock [t C ha^{-1}]	Reference
Short grass pasture	CH	1800-2050	0-20	84	Risch et al., 2008
Tall grass pasture	CH	1800-2050	0-20	66	Risch et al., 2008
Mountain pine forest	CH	1800-2050	0-20	91	Risch et al., 2008
Mixed conifer forest	CH	1800-2050	0-20	54	Risch et al., 2008
Stone pine, larch forest on acidic soil	CH	1800-2050	0-20	29	Risch et al., 2008
Hay meadow	A	1820-2000	0-10	35	Meyer et al., 2012
Pasture	A	1820-2000	0-10	40	Meyer et al., 2012
Abandoned grassland (25 yrs ago)	A	1820-2000	0-10	48	Meyer et al., 2012
Hay meadow	I	1790-1890	0-10	43	Meyer et al., 2012
Pasture	I	1790-1890	0-10	47	Meyer et al., 2012
Abandoned grassland (10 yrs ago)	I	1790-1890	0-10	45	Meyer et al., 2012
Meadow	I	600	0-30	153	Alberti et al., 2008
Mixed ash and sycamore forest (75 yrs)	I	600	0-30	107	Alberti et al., 2008

3.4.2 Different influences of SOC concentration and bulk density on SOC stock during green alder establishment

The different patterns of the SOC stock development along the analyzed chronosequences found in the literature (Chap. 3.4.1) and in the present study (Abbildung 11) illustrate that a closer examination of the SOC stock equation (Eq. 1) is needed. This could be achieved by testing the influence of the independent variables SOC concentration and soil bulk density on the dependent variable, namely SOC stock. The relationship between those variables could explain the SOC stocks (Abbildung 11) found in this study in more detail than just by land cover change. Possible mechanisms are discussed in this section.

In general, the conversion of grassland to woodland leads to a higher input of organic material into the soil due to the increased net primary production and slower carbon decomposition (Jackson et al., 2007; Bühlmann et al., 2016). Studies showed that an increased amount of soil organic matter results in a lower bulk density (Périé and Ouimet, 2008; Caviezel et al., 2014). This negative correlation can be confirmed by our results of the relationship between the SOC concentration and the bulk density ($R > -0.82$; $p < 0.05$). Hence, the two independent parameters of the SOC equation act contrary and, depending on the site conditions (such as vegetation type, vegetation age, soil fauna community and soil texture), influence the SOC stock differently. In the present study, the SOC concentration and the SOC stock correlated positively at a moderate level and the bulk density and the SOC stock showed a partly significant moderate relationship with a negative correlation, surprisingly.

To test how the two independent variables influence the SOC stock, the values of the SOC stock and those of the inversed bulk density were independently z-transformed by age class. In a next step, the linear regression coefficients (a) were plotted against the 1:1 line in the scatter plot between “inverse bulk density” and “SOC concentration” (Abbildung 14). The analysis revealed an age-dependent relationship between these two variables and hence each of these variables influences differently the SOC stock during shrub encroachment. While the regression coefficients *a* of the age classes GA15, GA25 and GA40 were lower than 1.0, they were considerably higher than 1.0 for the pasture sites and GA90. This means that the SOC stock is more influenced by the bulk density than the SOC concentration during the first stages of green alder establishment. Contrarily, the SOC concentration dominates the SOC stocks on pasture grassland and mature green alder stands.

Based on the different relationships between the SOC concentration and the bulk density of different stages of encroachment identified above, we assume that the main source of the SOC at the pasture sites are the root systems of the grass species consisting of fine roots (Jackson et al., 1996; Caviezel et al., 2014) that are able to penetrate through the mineral phases without a lasting effect on loosening the soil. Thus, the SOC stock of long-time used pastures is dominated by the SOC concentration (Abbildung 14). By turning these pastures into green alder shrubland, during the first stage of the bush establishment, the input of litter material and the changed rooting pattern lead to a loosening of the soil, which is measurable by a lower bulk density compared to the soil of pastures (Caviezel et al., 2014) (Abbildung 9). The bulk density, therefore, controls the SOC stock more than the SOC concentration during the first years of shrub growth resulting in a decrease of the SOC stock and negative sequestration rates (Abbildung 11, Abbildung 14). However, due to the decomposition of SOM and a possible physical weathering of SOM (see chapter 3.4.3) a larger portion of the SOC is associated with the S+C fraction (Abbildung 13) at mature green alder stands, which results in a higher bulk SOC concentration beside the still high OM input. Thus, the SOC stock is again more affected by the SOC concentration than by the bulk density (Abbildung 14).

The comparison between the SOC concentration and the soil bulk density and their changing influence on the SOC stock show how soil parameters can change in different ways during processes such as shrub encroachment and hence, it illustrates well the complexity of the soil system especially during the time of transition (Caviezel et al., 2014). The influence of the litter input on secondary parameters like bulk density, further, demonstrates that during system changes the biological perspective of the SOC stock which is the result of C-input and C-output (Vesterdal et al., 2013) should be supplemented by pedo-ecological reasoning to explain SOC stock changes in more detail.

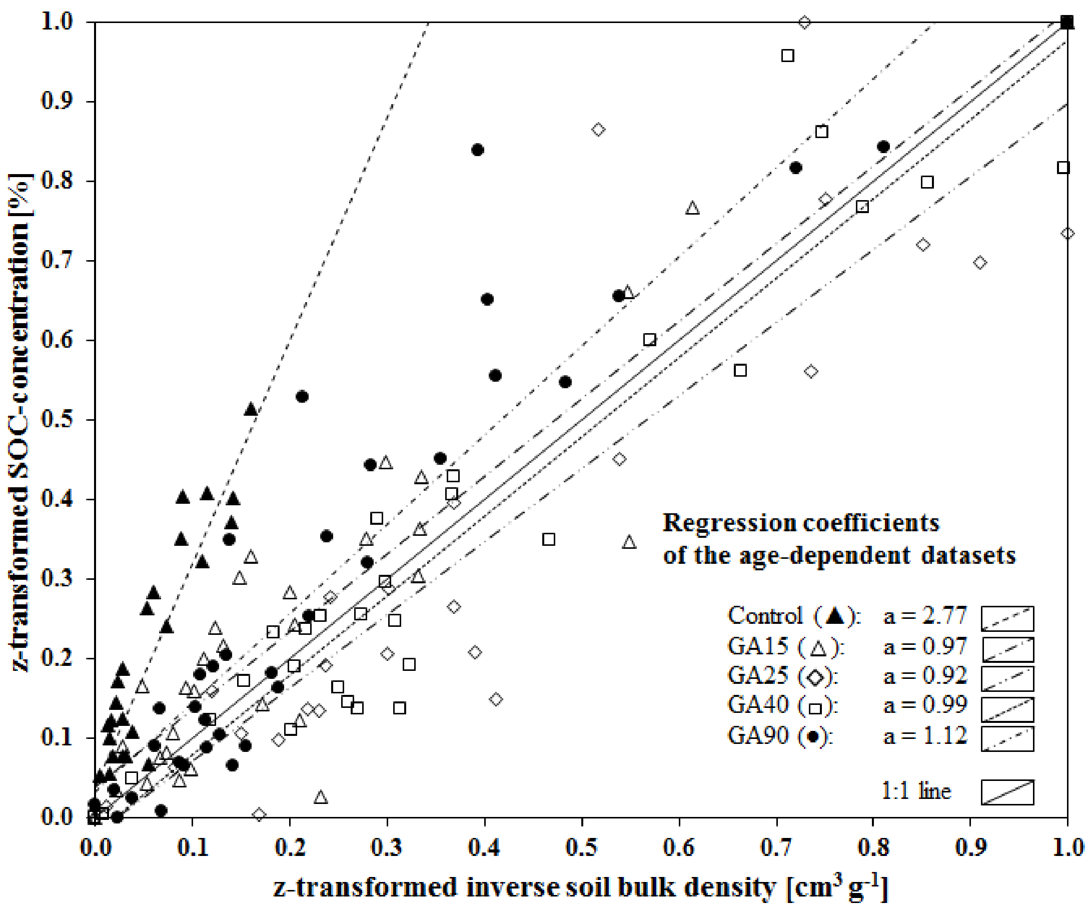

Abbildung 14: Relationship between the z-transformed data of the inverse bulk density [$cm^3\ g^{-1}$] and the SOC concentration [%]. The data was first classified by the age class before the transformation was performed.

3.4.3 Consequences of green alder encroachment on SOC stability and processes on landscape level

Green alder bushes boost the labile fraction of SOC stocks in subalpine environments

For all tested age classes, the POM fraction forms > 50 % of the SOC stock (0-30 cm) (Tabelle 4). The highest amount of POM material (approximately 115 t C ha^{-1}) was stored at the oldest green alder stands. This amount alone is comparable with the given total SOC stocks in the literature (Tabelle 5). However, the high ratio of POM material at all tested sites (incl. pasture sites) of the subalpine catchment (Tabelle 4) is consistent with the findings of Leifeld et al. (2009) and Budge et al. (2010). They found a positive correlation between the amount of fresh and labile organic material and the elevation due to the decrease of the biological activity at higher elevation (Leifeld et al., 2009; Sjögersten et al., 2011).

Another argumentation for the high SOC stock at GA90 might be the unintended sampling of organic layer material due to the difficult separation of the organic layer and the mineral soil at the high productive green alder stands. According to the supplementary material of Bühlmann et al. (2016), the litter of green alder shows a carbon concentration of 525 mg g^{-1}. The present study however measured a median SOC concentration of 271 mg g^{-1} (Q1: 221 mg g^{-1}, Q3: 355 mg g^{-1}) in the upper 5 cm of the sampled mineral soil. Thus, we can exclude that exclusively organic material was sampled. It appears that the significant increase of the SOC stock of the mineral soil which is mainly explained by the accumulation of POM material (Tabelle 4) is driven by the high biomass production (Wiedmer and Senn-Irlet, 2006), the fast expansion growth (Wiedmer and Senn-Irlet, 2006), the induced change of the site conditions towards a shady (Bühlmann et al., 2016) and hence cooler and more moist environment, which hampers the decomposition of the litter and its residues (Jackson et al., 2007) (Abbildung 9). Based on this, we suppose that the incorporation of organic material by the biological activity is hampered. However, the average POM mass showed a significant increase between Control (120 mg g^{-1} soil) and GA90 (248 mg g^{-1} soil) within the top 30 cm of the soil. We therefore suppose that the higher soil porosity (Caviezel et al., 2014) favors the vertical transportation of organic material by water because the amount of root biomass (< 2mm) cannot be contemplable for the increase of the POM mass due to its decrease during shrub establishment (Caviezel et al., 2014). Hence, landcover change by *A. viridis* encroachment on subalpine pastures significantly increases the mass, the carbon concentration (Abbildung 13) as well as the stock (Tabelle 4) within the already dominant POM fraction and therefore the lability of the SOC in the mineral soil (0-30 cm).

Effects of green alder encroachment on SOC stability

In the analyzed soil profiles in the Unteralptal, the POM fraction as dominant SOC pool increases with duration of shrub cover, while the SOC in the HF fraction declines (Abbildung 12; Abbildung 13). The analysis of the SOC quality, which is the aim of the study, implicates that the stabilization of the soil organic material in larger aggregates (> 63 µm) is retarded under an established shrub community dominated by green alder. These changes are possibly linked to the strong pH dependency of the soil carbon turnover rate (Leifeld et al., 2013) and the formation of biogenic aggregates (> 63 µm) which occurs at soil reaction pH > 5.0 (von Lützow et al., 2008). In our study, the pH value (1:2.5 0.01 M $CaCl_2$) was always below pH 5.0 and further dropped significantly from 4.2 at the grassy sites to 3.1 at the oldest green alder sites (Caviezel et al., 2014). These findings are consistent with the results of Poeplau and Don (2013), Guidi et al. (2014a) and Gabarrón-Galeote et al. (2015), who also observed a decreased physical SOC stability after the turning of grassland into forest land.

Other reasons for the low aggregate stability could be the change of the microclimate with reduced solar radiation and soil temperature due to a denser canopy cover at the green alder stands (Abbildung 9). This induces unfavorable conditions for the decomposer community with regard to the biological activity and, thus, the biological-induced aggregation (Sollins et al., 1996; Six et al., 1998; Nosetto et al., 2005; Creamer et al., 2011; Hiltbrunner et al., 2013). Furthermore, the decrease of the SOC concentration in the HF fraction (Abbildung 13) constitutes a hindrance for the development of more stable organo-mineral complexes. A possible reason for the decrease might be the destruction of soil aggregates by the prevailing physical weathering processes in mountainous ecosystems, namely temperature and frost wedging. These processes can become more distinguishable due to the measured increase of the pore volume during shrub encroachment (Caviezel et al., 2014) and hence the possible higher magnitude of temperature and the higher water storage capacity in the soil pores.

In the S+C fraction, the study showed an increase of the median SOC concentration (depth 0-30 cm: Control: $\tilde{x}$ = 74 mg g^{-1}, GA90: $\tilde{x}$ = 130 mg g^{-1}) and the median stock (depth 0-30 cm: Control: $\tilde{x}$ = 34 t C ha^{-1}, GA90: $\tilde{x}$ = 50 t C ha^{-1}) which starts at the beginning of the shrub encroachment (Abbildung 13; Tabelle 4). The above mentioned weathering by frost wedging might also be the reason for the destruction of larger organic material into components of the S+C fraction. This finding is comparable with the study of Gabarrón et al. (2015), who analyzed the effect of reforestation on abandoned cropland on SOC patterns in the Baetic mountains of Spain. However, the extraction of the S+C fraction by the physical separation technique of Zimmermann et al. (2007b) and the measurement of its C concentration hardly provide an indication for the stability of the SOC in the analyzed fraction. Hence, the method does not give any information about the location of the OM in the S+C fraction and consequently, the degree of the SOC stabilization. Therefore, and due to the circumstance, that the formation of organo-mineral complexes is pH dependent, it is questionable, whether the majority of the SOC is bounded to the mineral phase of the S+C fraction.

Overall, the observed trends of the SOC in the POM, HF and S+C fractions during the establishment of green alder shrub vegetation confirm the apprehension of Tasser et al. (2005), Sjögersten et al. (2011) and Guidi et al. (2014a) that due to the higher amount of labile and unprotected SOM, the vulnerability of the SOC will be increased by converting grassland into wooded land, especially green alder shrubland with its high annual litter production (Wiedmer and Senn-Irlet, 2006; Bühlmann et al., 2016). Thus, the information about the composition and the stability of the SOC has to be taken into account when SOC stocks of developing vegetation types under land-use change are assessed (John et al., 2005).

3.4.4 The SOC in the mountainous system – possible changes and risks due to shrub encroachment and climate change

The present study showed very strong and significant relationships between the POM mass and the soil stability parameters like bulk density, torsional shear strength and penetration resistance. Without testing the causality, we assume that the increased amount of POM material by green alder encroachment favor the swelling and shrinking processes of the soil matrix which increase porosity of the soil phases (see chapter 3.4.4). This leads to a higher water infiltration capacity and a higher saturation level of the soil under green alder stands, which was demonstrated by Caviezel et al. (2014). Consequently, the tendency of higher water percolation and less internal soil friction may favor the risk of soil moving processes.

Besides the decomposition of organic carbon, the leaching of dissolved organic carbon (DOC) can lower the SOC pool (Vesterdal et al., 2013). There is consequently a pathway for carbon from the terrestrial to the aquatic system. The values of the present study indicated significant increases of the DOC concentration (Abbildung 12) and are consistent with concentration in forest soils of previous studies (McDowell and Likens, 1988). According to Pregitzer et al. (2004) and Bühlmann et al. (2016), the N-saturation in the soil stimulates the production of DOC due to proton production and consequently, the lowering of pH value. For the green alder stands, the present study was able to demonstrate the negative relationship between pH value and DOC concentration which became stronger with increasing bush age (see chapter 3.4.4). In addition, the risk of leaching DOC is encreased by the enhanced water percolation through larger soil pores (Caviezel et al., 2014). Thus, the present study agrees with the findings of Mueller et al. (2016) that during rainfall events more soluble carbon from green alder sites can be transported to water bodies. Thus, a doubling of the DOC concentration (Abbildung 12) during the establishment of green alder bushes can affect aquatic systems like springs, draining ditches and ponds with consequences for their primary production and the food web within these sensitive habitats (Jones, 1992; Solomon et al., 2015).

The interaction of green alder with the surrounding environment might be modified as a consequence of climate change. Models predict increased temperature and more moist winters and heavy rainfall events during summer for the Alpine region in the next decades (Beniston, 2006). The changing climate can influence the geomorphic process domain and hence the SOC pool. Processes which affect the SOC can be the transportation of POM mass and thus POM-C by avalanches (Newesely et al., 2000; Caviezel et al., 2010; Meusburger and Alewell, 2014) or the movement of particular or dissolved organic carbon into the aquatic system by runoff (Caviezel et al., 2014) as well as soil creeping (Caviezel et al., 2014). Due to the increased frequency of droughts during warmer and drier summers, the risk of wild fires may also increase (Schumacher and Bugmann, 2006). Consequently, the combustion of the freely available SOC and the subsequent release of carbon dioxide into the atmosphere should be taken into account when assessing the environmental impact of green alder in the Alps.

3.5 Conclusion

This study focused on quantitative and qualitative patterns of the SOC and its interactions with additionally measured soil parameters. The results illustrate that the vegetation shift from subalpine pasture land to green alder shrubland substantially affects the soil system in the top

30 cm. During the establishment of the green alder vegetation type, the SOC stock in the mineral phases initially decreases from 100 t C ha^{-1} to 81 t C ha^{-1} and subsequently ends up in a significantly higher SOC stock (174 t C ha^{-1}) under well-established green alder stands. According to our results, the taken SOC stock of 69 t C ha^{-1} in the National Inventory Report therefore reflects, at the most, the situation for younger green alder stands and it is not representative for mature green alder stands.

Due to the significant increase in the particulate organic matter fraction (POM) and the depletion in the sand and aggregate fraction (HF), green alder encroachment leads to a lowering of the SOC stability. It is unclear whether the rise of the SOC in the silt and clay fraction (S+C) comes along with more stable carbon in the S+C fraction. Further, the change of the SOC properties induced by the ingrowth of *A. viridis* shrubs may influence the geomorphic and geochemical process domains.

Overall, the study clearly shows the difficult comparability of the green alder ecosystem with other mountainous ecosystems concerning soil organic carbon properties. Due to this and the increasing role of green alder shrub forests in mountainous regions, we therefore suggest to study the green alder shrub system as an independent vegetation type in the future to ensure i) the knowledge of the system processes and ii) the establishment of robust and reliable datasets concerning soil processes and nutrient pools for this vegetation type.

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KAPITEL 4

Evaluating the carbon sequestration potential of volcanic soils in south Iceland after birch afforestation

Der Fluss Ytri-Rangá fliesst westlich am Birkenbuschwald in "Hraunteigur" vorbei und hat mit seiner Barrierenfunktion in der Vegangenheit den Wald vor den von Norden herannahenden Sanddünen geschützt, weshalb der Wald eine Insel von natürlich gewachsenen, alten Birken im 900 km² grossen „Hekluskógar" um den Vulkan Hekla (im Hintergrund) darstellt. Aufgenommen von M. Hunziker am 10. Juli 2009.

Evaluating the carbon sequestration potential of volcanic soils in south Iceland after birch afforestation

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Abstract

Restoration activities are a strategy to bind atmospheric carbon in the terrestrial system for decades or hundreds of years and to enhance ecosystem services. Iceland's large areas of former vegetated and now degraded ecosystems show, therefore, a high potential to act as carbon sinks. Consequently, the ecological restoration of these ecosystems is part of mitigation programs and is financially supported by the Icelandic government. The aim of this study was to explore the change of the SOC pools during the establishment of birch vegetation and to estimate the SOC sequestration potential of afforestation by birch. With a chronosequence approach, different old afforested birch sites (15, 20, 25 and 50 years) were compared with sites of eroded land, naturally grown remnants of birch woodland and grasslands which were re-vegetated using fertilizer and grass seeds 50 years ago. Besides the estimation of the SOC stock, the taken soil samples were physically fractionated to characterize the quality of the SOC.

The results show that the degraded soils can potentially sequester additional 20 t C ha^{-1} (0-30 cm) to reach the SOC stock of naturally grown birch woodlands. During the establishment of afforested birch woodlands, the SOC stock (0-30 cm) continuously increased. After 50 years of birch growth, the SOC stock stated still on a significantly lower level compared to the stock of naturally grown birch woodlands. By using the calculated SOC sequestration rates (between -0.53 and 0.70 t C ha^{-1} yr^{-1}), the SOC stock of original birch woodlands is reached within 85 years. The SOC fractionation revealed that most of the carbon was stored in the fraction < 63 µm. However, the POM fraction enriched most intensive during the growth of mountain birch stands (+ 12 t POM-C ha^{-1}). We, therefore, assume that some of this POM carbon which is part of the calculated SOC stock is released to the atmosphere during the stabilization with the mineral soil phases in the future. Further, the study clearly illustrates the limitations of the chronosequence approach given by the heterogeneous landscape evolution resulting in severely degraded soils. Therefore, we suggest using repeated plot measurements in the future.

Keywords: land restoration, afforestation, sequestration potential, SOC stock, SOC quality, soil fractionation

4.1 Introduction

4.1.1 SOC change due to land use and landcover change

The terrestrial system in terms of the carbon compartments of vegetation and soil can be used to sequester atmospheric carbon (Houghton et al., 2004; Sabine, 2006). In an environmental-political perspective, countries can therefore act carbon-beneficial and report sequestration to the UN Framework Convention on Climate Change in their national carbon budgets to lower the annual emission numbers. On national scale, there are actions like afforestation, restoration or conversion into more productive land uses in the sector of Agriculture, Forestry and Other Land Use (AFOLU) to balance the statewide carbon emissions (Smith et al., 2014). Depending on the applied action, the enhancement of carbon sequestration is performed within or between different land use categories (Penman et al., 2003). The aimed result is a net carbon increase in the biomass or soil reservoir by reducing the emissions and enhancing the carbon input (Guo and Gifford, 2002; Smith et al., 2014).

4.1.2 Iceland's soil carbon sequestration potential by land restoration

Icelandic government approved activities like revegetation, afforestation and wetland restoration in the 1990's (Aradottir and Arnalds, 2001; Ministry for the Environment, 2007; Sigurdsson and Snorrason, 2000). However, reclamation of land has been carried out for more than 100 years in Iceland in order to halt land degradation and soil erosion events (Crofts, 2011) due to human activities in combination with natural stress factors, such as volcanic eruptions or harsh climate since the settlement about 1100 years ago (Aradottir and Arnalds, 2001).

About 43000 km^2 of Iceland (~40%) of Iceland have been affected by (extremely) severe soil erosion (Arnalds et al., 2016). Consequently, about 120-500 Mt soil organic carbon (SOC) have been lost in the past (Óskarsson et al., 2004). At the present, approximately 50000 km^2 (~50 %) are covered by barren deserts or are classified as disturbed areas (Arnalds, 2000). According to Arnalds et al. (2013), these landscapes are characterized by limited vegetation cover on vitric soil types resulting in low biomass production and low SOC stocks (Óskarsson et al., 2004). Regarding the carbon reservoir of Icelandic soil, Vitrisols or Leptosols which are the typical soil types of the deserts contain less than 45 t C ha^{-1} (Óskarsson et al., 2004). On the other hand, regarding dryland systems woodlands and species-rich heathlands form the un-disturbed ecosystem type (Aradóttir et al., 1992). Fertile Brown Andosol is the typical soil type of these ecosystems and it is found on 13360 km^2 (Óskarsson et al., 2004). Andosols have a tendency to accumulate higher quantities of SOC than other soil types due to the cover of SOC enriched surface horizons by volcanic ejecta, the andic properties and the formation of organo-mineral complexes with allophane and ferryhidrite as clay minerals (Dahlgren et al., 2004; McDaniel et al., 2012; Delmelle et al., 2015; Arnalds, 2015d). Hence, the SOC stock of Brown Andosol soils is estimated to be 227 t C ha^{-1} (Óskarsson et al., 2004).

Based on this, the severely degraded landscapes exhibit a high carbon sequestration potential by turning them again into productive and fertile landscapes (Arnalds et al., 2000; Ágústsdóttir, 2004; Lal, 2009). Therefore, the reclamation of degraded land is of high importance and is in consequence financially supported by the Icelandic government (Ministry for the Environment, 2007). A second aspect of reclaiming degraded land is the recovery or the enhancement of the ecosystem services like regaining of farm land, land protection against soil erosion or public recreation (Aradóttir et al., 2013). As an example the large-scale project called *Hekluskógar* was established in South Iceland in 2007 with the aim of restoring resilient birch woodlands on about 900 km^2 in the vicinity of Mount Hekla with continuous volcanic hazards (Aradóttir, 2007).

4.1.3 Assessment of SOC change in Iceland

Since the late 1990's, several field studies have estimated the effect of land reclamation with regards to the SOC development. For this, the widely applied method of calculating SOC stocks by Alde et al. (2006a) was used (Snorrason, 2010). The widely used technique to estimate the effect of land reclamation concerning SOC sequestration is to divide the difference of the SOC stock between two SOC inventories in time by the number of years between the two inventories (Aalde et al., 2006a; Hellsing et al., 2016). Hence, the Icelandic National Inventory Report uses a country specific removal factor of 0.51 t C ha^{-1} yr^{-1} for soils during the conversion of severely degraded land ("Other Land") to forest land or grassland (Hellsing et al., 2016). This is based on Icelandic field studies which found an increase of the SOC stock and therefore assigned a positive effect to the reclamation efforts regarding the sequestration of carbon in the soil (Aradóttir et al., 2000; Snorrason et al., 2002; Ritter, 2007; Bjarnadottir, 2009; Kolka-Jónsson, 2011; Arnalds et al., 2013). However, the establishing vegetation community passes through different development stages and it is further believed that SOC sequestration can reach a saturation level (Smith et al., 1997; Stewart et al., 2007). Hence, the sequestration rate is not linear during the establishment of the new vegetation cover and the achievement of the new SOC stock equilibrium (Six et al., 2002b). In order to consider these patterns, the development of the SOC stock and the SOC sequestration rates need to be recorded with a higher temporal resolution to estimate the sequestration potential of re-vegetated land instead of using SOC stocks values of only two inventories. Kolka-Jónsson (2011) only studied the SOC sequestration under different old afforested mountain birch stands.

Soil organic matter consists of a heterogeneous matrix with respect its physical protection and chemical structure (Schmidt et al., 2011). Hence, these different organic compounds differ in decomposability and turnover times (von Lützow et al., 2008). During land-use changes, the composition of the SOC changes (Poeplau and Don, 2013). Several recently published studies showed that as a consequence of revegetation or afforestation the more likely labile SOC pool which consists mainly of particulate organic matter (POM) simultaneously rises with the increase of the total SOC in the mineral soil during the establishment of systems with a higher net primary production rate as before (Guidi et al., 2014a; Gabarrón-Galeote et al., 2015; Trigalet et al., 2016; Hunziker et al., 2017). It is unclear, how afforestation with mountain birch affects the SOC quality of formerly severely degraded soils.

The enhancement of the organic carbon pool of well-drained soils can be undertaken in different ways by vegetation change. In Iceland, revegetation of severely degraded land (Aradóttir et al., 2000; Arnalds et al., 2013), the afforestation by different types of tree species on heathland (Ritter, 2007) as well as on grazed land (Snorrason et al., 2002) and the natural, unmanaged vegetation succession and soil development in due to glacier retreat (Vilmundardóttir et al., 2015) are conversion types that have been already studied. As it is shown, the afforestation with the only native tree species *Betula pubescens* Ehrh. *ssp. czerepanovii* on the soil organic carbon has not been studied yet.

The present study was part of the *CarbBirch* project in Iceland (Halldórsson et al., 2011) which was launched in 2008 and involved two of the five *CarbBirch* study sites. The main goal of *CarbBirch* was to study the ecological impact of the restoration activities in *Hekluskógar*. The present study aims at characterizing the long-term carbon sequestration potential of the implemented afforestation efforts with mountain birch on severely degraded soils. For this, we compared the SOC stocks and the SOC quality by physical soil fractionation at mountain birch stands of different ages to those of severely degraded and barren areas, reclaimed grasslands and natural old growth birch woodlands.

4.2 Material and methods

4.2.1 Study setup

The study area is in the vicinity of the volcano Mount Hekla (Abbildung 15; A). Due to the unsustainable land use and volcanic activities, most of this area has been affected by erosion (mostly in form of huge sandstorms) (Crofts, 2011; Arnalds et al., 2016). Hence, the landscape is characterized by sandy deserts (Arnalds et al., 2016). The parent material of the soils can consists of material of capped soil profiles, aeolian deposition, material of lava fields or glacial till (Dugmore et al., 2009; Thorarinsdottir and Arnalds, 2012). Reclamation activities have been carried out on these surfaces during the last decades (Halldórsson et al., 2011).

The afforested woodland area "*Gunnlaugsskógur*" is located approximately 1 km north of the Icelandic Soil Conservation Service Headquarters at Gunnarsholt (Abbildung 15; C). In 1926, the eroded area was protected from sheep grazing by fencing. After the stabilization of the ground surface and the fertilization of the soil, birch seeds were put out on small plots in 1939 and 1945. Additionally, in 1945 birch seedlings, resulting from the activity in 1939, were transplanted on a nearby lava field. However, most of the present birch area at *Gunnlaugsskógur* was naturally generated by seed production of the formerly established birch, resulting in new stands of different ages (Aradóttir, 1991; Aradottir and Arnalds, 2001). In the *CarbBirch* project, the age of the afforested birch sites was determined by dendrochronology. The mean ages of the sampled birch plots were 15, 20, 25 and 50 years (Birch15, Birch20, Birch25, Birch50), respectively. In addition to the birch plots, soil samples were taken from three severely degraded sites with barren surfaces and from three revegetated sites with grass vegetation north of *Gunnlaugsskógur* (Abbildung 15; D). In the present study, the severely degraded and eroded sites (Barren Land) act as the stage before any restoration activity has begun. The grassland sites (Grass50) were protected against sheep grazing by fencing and then revegetated by using fertilizers and grass seeds about 50 years ago. The grassland sites are not used for hay production. The soils of these sites can however contain organic carbon due to the burial of carbon rich surface horizons with deposited material which can further contain carbon (Arnalds, 2010; Arnalds et al., 2013). This has to be considered when assessing carbon sequestration by restoration programs on disturbed landscapes. Due to the same age of Birch50 and Grass50, the two different reclamation types can be directly compared. The different old birch sites were further compared to a naturally grown birch woodland which is located at "*Hraunteigur*" (Abbildung 15; B). This area was protected against sand encroachment by two streams but was subjected to deposition of large amounts of dust and periodic tephra fallout. Thus, it holds remnants of the original mountain birch woodlands (Birchnat), which covered large areas in the vicinity of Mount Hekla (Árnason, 1958) in the past. Due to the long term continuous vegetation cover subjected to large scale sediment deposition, this area has accumulated thick soils with a soil depth of more than 2 m (Helgason, 1899; Kolka-Jónsson, 2011).

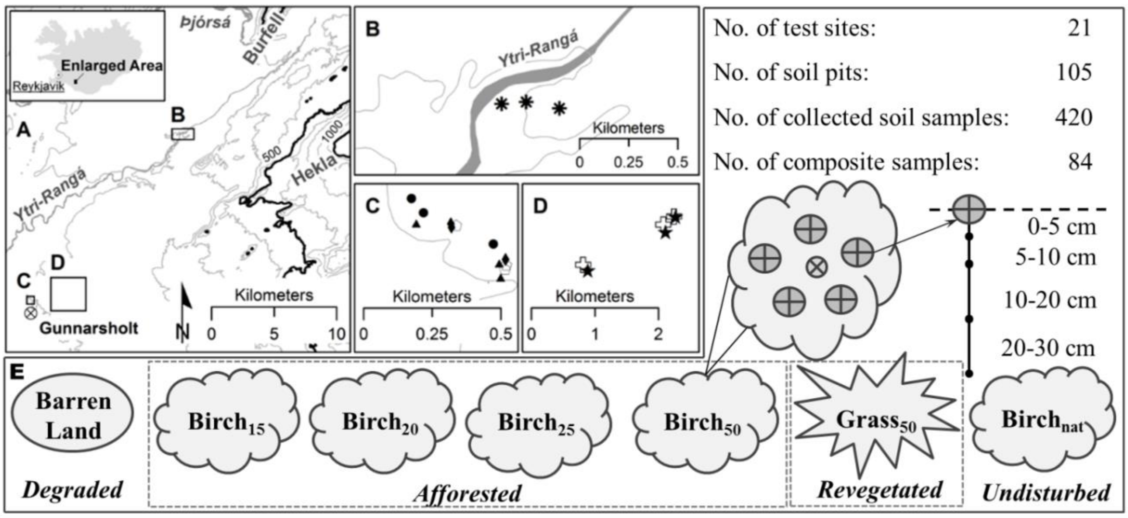

Abbildung 15: The topological map (equidistance = 100 m) showing the study area and Gunnarsholt (crossed cycle) in the south of Iceland (A). The locations of the natural grown birch woodland (B; asterisks) and the afforested (C; B15: circles, B20: triangles; B25: pentagons; B50: diamonds) and degraded (crosses) as well the revegetated (stars) test sites (D) are shown in more detail. The sampling scheme

figures the age and vegetation characteristics of the different study sites and the applied soil sampling setup (E).

Field sampling was carried out in summer 2011. Each of the above described landcover type and age categories was represented by three test sites resulting in 21 sampling sites (Abbildung 15; E). After removing the litter layer, the top 30 cm of the mineral soil was sampled which is the common depth for SOC stock inventories (Aalde et al., 2006b; Snorrason, 2010). A second argument for the use of the mentioned sampling depth is that the top 30 cm contains most of the belowground living root biomass at grassland and birch areas, respectively, and thus comprises the belowground organic carbon sources (Snorrason et al., 2002; Bjarnadottir et al., 2007; Hunziker et al., 2014).

At each site, five soil pits were randomly placed. At the woody sites, sampling occurred within one half of the crown diameter of a dominant mountain birch (*Betula pubescens* Ehrh. *ssp. czerepanovii*) tree. Per pit, the soil was sampled with a cylindric metal core (Eijkelkamp Soil & Water, Giesbeek) of 100 cm^3 volume and 5 cm in diameter at given soil intervals (0-5, 5-10, 10-20 and 20-30 cm). The study is focused on the SOC change between different landcover types as well different old birch stands and less on the spatial heterogeneity at plant scale. Thus and in order to make the dataset more robust, at each sampling site the five sub-samples per depth interval were immediately mixed in order to form one composite sample. Thus, each depth interval per category was therefore represented by three composite samples (Abbildung 15).

4.2.2 Soil treatments in the laboratory

Determining common properties for volcanic soils

All 84 composite soil samples were dried at 40 °C until a constant weight was reached. The weight [g], the volume [cm^3] and the bulk density [g cm^{-3}] of the fine earth (< 2 mm) were determined by dry sieving and water displacement of the coarse material (> 2mm). Soil reaction (pH value, [-]) was determined in water (1:2.5) and potassium chloride (1:2.5 0.01 M KCl) (FAL, 1996). Acid ammonium oxalate extractable Al, Fe and Si and pyrophosphate extractable Al and Fe were measured with an ICP device after the method of Blakemore et al. (1987). The concentrations [%] of the volcanic clay minerals allophane and ferrihydrite were estimated by multiplying the Si_{ox} concentration by 6 and the Fe_{ox} concentration by 1.7, respectively (Parfitt and Childs, 1988; Parfitt, 1990). The allophane and ferrihydrite concents were then summed up to determine the clay content [%] deriving from the oxalate extraction which is a typical measure for texture analysis in volcanic soils (Arnalds, 2015d).

SOC fractionation

The responses of SOC stability to land-use changes can be investigated through physical fractionation methods, which allow to investigate the organization of OM within the soil matrix (Cambardella and Elliott, 1992; Six et al., 1998; Poeplau and Don, 2013). We therefore slightly modified the separation procedure presented by Zimmermann et al. (2007b) and used the disaggregation, the particle size separation and the density fractionation to separate the SOC into the given functional groups (von Lützow et al., 2007). The fractionation procedure was performed with the material of the fine earth (< 2 mm) of the 180 composite samples. Initially, the samples were dispersed by an ultrasound treatment (22 J ml^{-1}) in 150 ml deionized water to retrieve only primary organo-mineral complexes (Christensen, 2001). Afterwards, the samples were wet sieved at 63 microns to separate the stable sand-sized aggregates and the un-protected particulate organic matter from the material < 63 microns. Subsequently, the particulate organic material (POM) was separated from the denser organic material in the mineral-associated sand and aggregate fraction (heavy fraction; HF) by density fractionation (1.8 g cm^{-3}, SPT from Sometu) on the soil material (> 63 microns). After separation, both fractions were washed with deionized water until the electrical conductivity of the rinse water reached < 50 µS (Wagai et al., 2008). In some cases, the volcanic pumice material exhibited a density < 1.8 g cm^{-3} and, therefore, contaminated the POM fraction. To solve this problem, we applied the electrostatic attraction technique explained by Kaiser et al. (2009). The pumice material (< 1.8 g cm^{-3}) was transferred to the HF fraction. The material which is smaller than 63 microns represents the SOC pool of the silt and clay size fraction (< 63 µm) which can also contain aggregates consisting of volcanic clay minerals. Further, after settling time, a sample of the suspension (< 63 microns) was taken, filtered with a 0.45 microns filter and analyzed for its dissolved organic carbon content (DOC). The value of the DOC concentration was used as an indication for the ability of the studied soils to leach dissolved organic carbon. Compared to Zimmermann et al. (2007b), the present did not conducted the oxidation with sodium hypochlorite (NaOCl).

All samples of the bulk soil (< 2 mm) and the POM, HF and < 63 µm fractions were ball-milled and then analyzed for the organic carbon content [%] by the dry combustion technique (Leco CN 628 Elemental Determinator). The DOC content [mg l^{-1}] was measured by the combustion analytic oxidation method (TOC-5000A, Shimadzu).

4.2.3 Calculations and statistical analysis

General statistical practices

The statistical analysis was performed with the software Statistical Package for the Social Sciences (SPSS; IBM, Vers. 22). Due to the small amount of replicates (N = 3) per site and depth class, the median value and the range of the values were chosen to describe the results. The nonparametric Mann-Whitney U test was used to detect significant ($p < 0.05$) differences between different sites within the same depth interval.

The estimation of the SOC stock and the sequestration rate

The amount of soil organic carbon, that is stored in a given soil profile, is defined as the SOC stock and is given in tons per hectare. According to Ellert et al. (2008) and Rodeghiero et al. (2009), the SOC stock (SOC_{stock}; [t C ha^{-1}]) is a function of the soil's carbon concentration (SOC_{conc}; [mg g^{-1}]), the bulk density (BD; [g cm^{-3}]) of the fine earth (< 2 mm) and the investigated soil depth (l; [cm]). The conversion factor between the units is 100.

$$SOC_{stock} = SOC_{conc} \times BD \times l \times 100 \qquad \text{(Eq. 1)}$$

The SOC stocks which are stored in the investigated SOC fractions were calculated after Poeplau and Don (2013) and Guidi et al. (2014a). To estimate the annual sequestration rate, we applied the widely-used SOC stock-change method on the calculated SOC stocks (0-30 cm). The annual change is estimated by dividing the difference in SOC stock at two points in time by the number of years between the two SOC inventories (Aalde et al., 2006a). The SOC stock of Barren Land was used as reference SOC stock for evaluating the carbon sequestration of revegetation and afforestation. Further, the annual SOC stock changes were also calculated between the different old afforested birch categories (Birch15, Birch20, Birch25 and Birch50). Concerning soil organic carbon sequestration, the main goal of the establishment of vegetation cover during restoration is to promotes the production of living biomass and the incorporation of the organic carbon into the soil at which the aboveground part of woody biomass plays a key role (Jackson et al., 1996; Aradóttir et al., 2000; Snorrason et al., 2002). Hence, the SOC in the uppermost soil layer is mostly affected by the landcover change due to the vertical transfer of dead organic material from the soil surface. The study also examined the annual sequestration rates for 0-10 cm because of the given reason and to suppress a possible influence of the stocks in 10-30 cm by SOC which does not derive from reclamation.

4.3 Results

4.3.1 Physical, chemical and morphological characteristics of the sampled soil intervals

Differences between the studied category types were found concerning the volume of the gravel material (> 2 mm) (Tabelle 6). The volume of the gravel material significantly ($p<0.05$) differed between Barren Land and Birchnat as well as between Grass50 and Birch50. The calculated bulk densities of the fine earth material were within the range of 0.3 to 0.8 [g cm^{-3}] of Icelandic Andosols (Arnalds, 2008). In all sampled soil intervals, the clay content was dominated by allophane and was especially high in soils of Barren Land and Grass50 (Tabelle 7). Further, the minimum and maximum clay content was 9.04 % and 36.50 %, respectively, and the range between the first and third quartil was between 14.86 % and 20.30 % (Tabelle 7). The SOC concentration varied between 0.6 and 9.8 %. But the smallest C contents were not found at Barren Land as it was expected. At Barren Land, Grass50 and also at Birch 15, the SOC concentrations were higher in the deeper sampling intervals than in the depth interval of 0-5 cm (Tabelle 6). At Barren Land, Grass 50 and the afforested Birch sites, the C:N ratios of the soil mostly varied between 10 and 14. However, the ratio was considerable higher in the top 5 cm at the birch and grass sites which can be attributed to the presence of less decomposed organic matter deriving from woody material. This pattern was more distinct in the soils at Birchnat. These soils showed the highest C:N ratios with a maximum value of 19.2 (Tabelle 6). The Al_{pyr}:Al_{ox} and Fe_{pyr}:Fe_{ox} ratios which indicate that considerable amounts of Al and Fe are associated with metal-humus complexes generally showed a decline with soil depth at all vegetation types. Further, Birchnat and Birch50 showed the highest Al_{pyr}:Al_{ox} and Fe_{pyr}:Fe_{ox} ratios. At Barren Land and Grass50, the lowest ratios were found (Tabelle 7). Further, Barren Land showed the highest concentrations of volcanic clay minerals (allophane and ferrihydrite) (Tabelle 7).

All soil intervals showed relatively high pH-values (5.8-7.3) (H_2O) compared to the usually range of 5.5 to 6.5 for Andosols. The reason is the steady eolian input of un-weathered volcanic material which afterwards starts to weather and release ions (Arnalds, 2008). The soils of “Barren Land” showed significant ($p<0.05$) higher pH-values compared to those of the other tested categories (Tabelle 6) which is probably credited to the absence of vegetation at these sites which would reduce the pH-value by organic acids (Arnalds, 2008).

All sampled soil intervals fulfilled the andic properties ($(Al + ½Fe)_{ox} > 2$ %) and are therefore classified as Andosols after the IUSS Working Group WRB (2014) (Tabelle 7). According to the Icelandic soil classification (Arnalds, 2008), freely-drained soils under vegetation are termed as Andosols and soils of the deserts are classified as Vitrisols. The results of the present study confirmed that the sampled soils of the birch and grass stands are classified as Brown Andosols

(1-12% C and > 6 % allophane) (Tabelle 6, Tabelle 7). However, the severely degraded and un-vegetated sites (Barren Land) were also classified as Brown Andosols due to achieving the requirements (C- and allophane contents).

Tabelle 6: Values of the studied vegetation types and sampled soil intervals for common soil properties. The median value and the minimum and maximum values (in paranthesis) are given.

Type	Depth	Volume gravel (> 2 mm)	Bulk density (< 2 mm)	C content	C:N ratio	pH (H_2O)	pH (KCl)
	[cm]	[cm^3 100 cm^{-3}]	[g cm^{-3}]	[%]	[-]	[-]	[-]
Barren Land	0-5	11.8 (1.3; 14.5)	0.76 (0.64; 0.82)	1.7 (0.9; 2.9)	10.7 (9.9; 13.5)	7.0 (6.7; 7.1)	5.7 (5.4; 5.7)
	5-10	5.0 (0.5; 11.3)	0.65 (0.60; 0.82)	3.1 (0.9; 3.2)	13.3 (10.1; 14.8)	7.2 (7.0; 7.2)	5.8 (5.6; 5.8)
	10-20	0.5 (0.3; 9.3)	0.54 (0.49; 0.79)	1.7 (1.1; 2.4)	12.1 (10.6; 13.3)	7.2 (7.0; 7.2)	5.8 (5.7; 5.9)
	20-30	2.8 (0.5; 3.8)	0.48 (0.48; 0.56)	2.7 (2.2; 2.8)	13.5 (11.1; 14.5)	7.3 (6.8; 7.3)	5.9 (5.5; 5.9)
Birch15	0-5	6.8 (1.0; 8.3)	0.75 (0.66; 0.85)	2.1 (1.4; 2.4)	15.2 (14.4; 17.5)	6.1 (6.0; 6.4)	4.9 (4.8; 5.0)
	5-10	4.5 (1.3; 5.3)	0.87 (0.80; 0.89)	0.9 (0.9; 1.3)	11.5 (11.5; 12.7)	6.6 (6.3; 6.6)	5.2 (5.0; 5.2)
	10-20	3.0 (1.0; 4.8)	0.89 (0.69; 0.91)	1.1 (0.6; 2.0)	10.7 (10.5; 12.1)	6.8 (6.7; 6.8)	5.3 (5.2; 5.4)
	20-30	4.0 (0.0; 8.3)	0.76 (0.56; 0.90)	1.1 (0.4; 2.8)	11.1 (10.0; 11.9)	6.9 (6.8; 6.9)	5.5 (5.4; 5.5)
Birch20	0-5	1.5 (0.8; 4.3)	0.55 (0.47; 0.69)	2.9 (2.1; 5.0)	15.6 (15.6; 17.2)	6.1 (6.0; 6.2)	4.9 (4.9; 5.0)
	5-10	1.0 (0.5; 3.0)	0.79 (0.66; 0.89)	1.5 (0.8; 2.0)	12.0 (10.1; 13.7)	6.5 (6.4; 6.6)	5.1 (5.1; 5.3)
	10-20	1.0 (0.5; 3.3)	0.82 (0.69; 0.89)	1.1 (0.7; 1.7)	11.1 (10.1; 12.8)	6.7 (6.6; 6.9)	5.3 (5.2; 5.5)
	20-30	1.3 (0.3; 5.0)	0.91 (0.66; 0.95)	1.1 (0.8; 1.8)	11.2 (10.6; 14.7)	6.8 (6.8; 7.0)	5.3 (5.3; 5.6)
Birch25	0-5	2.3 (1.0; 8.0)	0.59 (0.44; 0.76)	3.4 (2.1; 5.5)	15.9 (14.1; 17.0)	6.1 (6.0; 6.3)	5.0 (5.0; 5.1)
	5-10	0.5 (0.4; 2.5)	0.77 (0.75; 0.89)	1.8 (1.0; 2.0)	12.2 (11.5; 13.3)	6.5 (6.5; 6.7)	5.2 (5.2; 5.2)
	10-20	3.0 (0.3; 3.0)	0.82 (0.80; 0.89)	1.1 (1.0; 1.5)	11.1 (10.9; 11.9)	6.7 (6.7; 6.7)	5.3 (5.2; 5.4)
	20-30	0.5 (0.1; 1.8)	0.79 (0.74; 0.90)	1.4 (1.0; 1.7)	11.0 (10.4; 11.3)	6.7 (6.7; 6.8)	5.3 (5.3; 5.4)

Fortsetzung

Type	Depth	Volume gravel (> 2 mm)	Bulk density (< 2 mm)	C content	C:N ratio	pH (H_2O)	pH (KCl)
	[cm]	[cm^3 100 cm^{-3}]	[g cm^{-3}]	[%]	[-]	[-]	[-]
Birch50	0-5	1.0 (1.0; 1.8)	0.44 (0.40; 0.49)	8.1 (5.5; 9.8)	18.6 (16.9; 20.9)	5.8 (5.8; 6.0)	4.8 (4.8; 4.8)
	5-10	0.8 (0.5; 2.5)	0.75 (0.68; 0.78)	1.9 (1.7; 2.4)	12.7 (12.5; 13.7)	6.3 (6.3; 6.4)	5.0 (5.0; 5.0)
	10-20	3.0 (1.3; 4.5)	0.78 (0.72; 0.84)	1.5 (1.1; 1.8)	11.8 (10.9; 12.2)	6.5 (6.1; 6.5)	5.1 (5.1; 5.2)
	20-30	0.3 (0.1; 1.1)	0.88 (0.85; 0.90)	1.1 (0.9; 1.3)	10.7 (10.6; 11.9)	6.6 (6.6; 6.9)	5.2 (5.1; 5.3)
Grass50	0-5	6.3 (6.3; 8.8)	0.72 (0.68; 0.73)	2.5 (2.5; 2.8)	12.6 (12.5; 13.0)	6.4 (6.3; 6.5)	5.1 (5.0; 5.1)
	5-10	7.5 (5.0; 7.5)	0.85 (0.71; 0.86)	2.0 (1.2; 2.3)	11.2 (10.7; 11.4)	6.7 (6.7; 6.7)	5.4 (5.2; 5.4)
	10-20	5.5 (3.8; 13.8)	0.63 (0.63; 0.75)	2.6 (1.8; 3.4)	10.9 (10.5; 11.4)	6.8 (6.8; 6.9)	5.5 (5.5; 5.5)
	20-30	1.5 (0.2; 7.5)	0.63 (0.61; 0.76)	3.4 (2.0; 3.4)	12.2 (12.1; 12.5)	7.0 (6.9; 7.1)	5.6 (5.5; 5.8)
Birchnat	0-5	0.5 (0.5; 0.5)	0.51 (0.46; 0.52)	6.3 (6.3; 6.5)	19.2 (19.0; 19.2)	6.0 (6.0; 6.2)	5.0 (5.0; 5.1)
	5-10	0.3 (0.1; 0.6)	0.68 (0.64; 0.68)	4.0 (3.3; 5.1)	16.3 (16.1; 17.7)	6.3 (6.3; 6.4)	5.1 (5.0; 5.1)
	10-20	0.1 (0.0; 0.1)	0.67 (0.64; 0.67)	2.4 (2.1; 3.4)	13.7 (13.2; 15.7)	6.5 (6.2; 6.6)	5.2 (5.0; 5.3)
	20-30	0.1 (0.0; 0.3)	0.72 (0.70; 0.76)	1.9 (1.8; 2.0)	12.6 (11.8; 12.8)	6.7 (6.7; 6.8)	5.3 (5.2; 5.4)

Tabelle 7: Volcanic soil properties of the studied vegetation types and sampled depth intervals. The median value and the minimum and maximum values (in paranthesis) are given.

Type	Depth	Al_{pyr}	Al_{pyr} : Al_{ox} ratio	Fe_{pyr}	Fe_{pyr} : Fe_{ox} ratio	$(Al + ½Fe)_{ox}$	Allophane	Allophane + Ferrhydrite Clay
	[cm]	[%]	[10^1, -]	[%]	[10^1, -]	[%]	[%]	[%]
Barren	0-5	0.20 (0.14; 0.27)	0.82 (0.72; 1.07)	0.17 (0.13; 0.23)	0.42 (0.39; 0.48)	4.50 (2.65; 6.65)	13.9 (8.2; 20.2)	20.8 (12.7; 30.3)
Land	5-10	0.26 (0.13; 0.27)	0.63 (0.60; 0.90)	0.22 (0.13; 0.24)	0.36 (0.34; 0.47)	7.46 (2.81; 7.71)	22.7 (9.3; 23.7)	33.9 (13.9; 34.8)
	10-20	0.16 (0.14; 0.17)	0.79 (0.54; 0.85)	0.18 (0.14; 0.19)	0.47 (0.35; 0.50)	3.90 (3.17; 5.85)	12.4 (10.4; 18.7)	18.6 (15.6; 27.8)
	20-30	0.24 (0.15; 0.28)	0.57 (0.44; 1.01)	0.23 (0.21; 0.29)	0.37 (0.35; 0.65)	6.51 (5.03; 7.31)	21.7 (15.2; 23.2)	31.8 (22.9; 33.8)
Birch15	0-5	0.26 (0.20; 0.26)	1.76 (1.55; 1.90)	0.21 (0.15; 0.21)	0.69 (0.55; 0.71)	2.82 (2.60; 3.01)	8.6 (8.2; 9.5)	13.6 (12.7; 14.6)
	5-10	0.17 (0.17; 0.19)	1.23 (1.13; 1.24)	0.14 (0.13; 0.15)	0.47 (0.41; 0.50)	2.96 (2.95; 3.05)	9.5 (9.4; 9.8)	14.8 (14.6; 14.8)
	10-20	0.19 (0.13; 0.27)	1.00 (0.95; 1.11)	0.17 (0.11; 0.25)	0.49 (0.36; 0.50)	3.37 (2.73; 5.35)	10.8 (9.5; 17.0)	16.5 (14.5; 25.7)
	20-30	0.18 (0.09; 0.31)	0.8 (0.75; 1.08)	0.15 (0.09; 0.36)	0.49 (0.34; 0.52)	3.26 (2.51; 7.59)	10.5 (8.8; 24.7)	15.9 (13.5; 36.5)
Birch20	0-5	0.33 (0.26; 0.36)	1.67 (1.44; 2.21)	0.28 (0.24; 0.41)	0.77 (0.64; 1.37)	3.15 (3.13; 4.49)	9.5 (8.9; 13.4)	14.9 (14.0; 20.8)
	5-10	0.21 (0.16; 0.26)	1.09 (0.95; 1.19)	0.18 (0.13; 0.25)	0.51 (0.44; 0.53)	3.52 (2.93; 5.09)	11.4 (9.5; 16.3)	17.3 (14.6; 24.4)
	10-20	0.17 (0.16; 0.20)	0.97 (0.85; 1.00)	0.16 (0.15; 0.21)	0.47 (0.46; 0.52)	3.56 (3.16; 4.44)	12.0 (11.1; 14.4)	17.9 (16.5; 21.5)
	20-30	0.19 (0.14; 0.19)	0.92 (0.69; 0.96)	0.19 (0.15; 0.23)	0.48 (0.47; 0.49)	3.99 (3.14; 5.11)	13.6 (11.0; 17.5)	20.3 (16.4; 25.6)
Birch25	0-5	0.32 (0.26; 0.38)	1.70 (1.64; 2.23)	0.32 (0.24; 0.42)	0.91 (0.73; 1.26)	3.37 (3.18; 3.65)	10.1 (10.0; 10.9)	15.6 (15.6; 17.0)
	5-10	0.23 (0.18; 0.25)	1.22 (1.09; 1.23)	0.20 (0.16; 0.22)	0.55 (0.48; 0.57)	3.76 (3.34; 3.94)	11.9 (11.3; 12.7)	18.1 (17.0; 19.2)
	10-20	0.18 (0.17; 0.22)	1.01 (0.92; 1.1)	0.16 (0.15; 0.19)	0.47 (0.41; 0.50)	3.81 (3.36; 3.98)	12.7 (11.6; 12.9)	19.0 (17.4; 19.5)
	20-30	0.22 (0.16; 0.24)	1.12 (0.89; 1.12)	0.20 (0.15; 0.22)	0.52 (0.42; 0.53)	3.92 (3.52; 4.23)	12.8 (11.8; 14.0)	19.3 (17.8; 21.1)

Fortsetzung

Type	Depth	Al_{pyr}	$Al_{pyr} : Al_{ox}$ ratio	Fe_{pyr}	$Fe_{pyr} : Fe_{ox}$ ratio	$(Al + ½Fe)_{ox}$	Allophane	Allophane + Ferrhydrite Clay
	[cm]	[%]	[10^1, -]	[%]	[10^1, -]	[%]	[%]	[%]
Birch50	0-5	0.52 (0.45; 0.58)	2.83 (2.81; 3.12)	0.65 (0.58; 0.72)	1.92 (1.86; 2.01)	3.57 (3.13; 3.65)	9.2 (8.4; 10.1)	15.1 (13.6; 16.2)
	5-10	0.30 (0.26; 0.34)	1.54 (1.26; 1.72)	0.28 (0.22; 0.31)	0.76 (0.55; 0.82)	3.89 (3.79; 4.08)	11.7 (11.5; 12.6)	18.0 (17.9; 19.4)
	10-20	0.24 (0.20; 0.25)	1.13 (1.13; 1.14)	0.21 (0.18; 0.22)	0.53 (0.50; 0.58)	4.08 (3.44; 4.20)	12.4 (11.5; 13.5)	18.6 (17.3; 20.5)
	20-30	0.19 (0.14; 0.19)	1.00 (0.86; 1.02)	0.18 (0.14; 0.18)	0.48 (0.43; 0.54)	3.55 (3.25; 3.74)	11.2 (11.0; 12.3)	16.8 (16.5; 18.6)
Grass50	0-5	0.29 (0.28; 0.29)	1.44 (1.33; 1.65)	0.24 (0.24; 0.26)	0.71 (0.66; 0.80)	3.62 (3.38; 4.03)	10.4 (10.0; 11.9)	16.2 (15.5; 18.1)
	5-10	0.24 (0.19; 0.25)	1.12 (1.09; 1.15)	0.20 (0.18; 0.25)	0.55 (0.53; 0.62)	4.10 (3.27; 4.30)	12.0 (10.3; 12.9)	18.5 (15.7; 19.6)
	10-20	0.31 (0.25; 0.34)	1.23 (1.23; 1.29)	0.25 (0.24; 0.32)	0.65 (0.58; 0.73)	4.67 (3.93; 4.82)	13.1 (12.0; 13.5)	20.3 (18.3; 21.1)
	20-30	0.26 (0.26; 0.32)	1.03 (1.00; 1.09)	0.30 (0.26; 0.33)	0.61 (0.58; 0.74)	4.77 (4.75; 5.41)	15.6 (13.5; 15.9)	23.1 (21.0; 24.3)
Birchnat	0-5	0.33 (0.30; 0.37)	2.81 (2.54; 2.93)	0.44 (0.42; 0.49)	2.18 (1.93; 2.21)	2.27 (2.14; 2.45)	5.9 (5.6; 6.5)	9.6 (9.0; 10.3)
	5-10	0.33 (0.32; 0.39)	2.34 (2.15; 2.66)	0.40 (0.39; 0.52)	1.68 (1.47; 2.05)	2.76 (2.52; 2.87)	7.4 (6.9; 7.9)	11.8 (10.9; 12.5)
	10-20	0.27 (0.24; 0.29)	1.49 (1.27; 1.51)	0.28 (0.25; 0.32)	0.88 (0.79; 1.06)	3.47 (3.33; 3.54)	10.2 (9.4; 10.3)	15.6 (14.6; 15.7)
	20-30	0.25 (0.24; 0.27)	1.20 (1.20; 1.34)	0.25 (0.24; 0.27)	0.74 (0.70; 0.83)	3.67 (3.63; 3.81)	10.9 (10.8; 11.5)	16.6 (16.4; 17.3)

4.3.2 Bulk SOC stocks

The SOC stock increased during the establishment of birch woodlands within the top 30 cm (Birch15 to Birch50) (Abbildung 16). The increase was highest in the top two sampling intervals and resulted in a significantly different SOC stocks between Birch15 and Birch 50. The 50 years old afforested sites had a considerably lower median SOC stock (0-30 cm) than the old-growth woodland (∆ 15 t ha^{-1}). Thus, the remnant birch woodland (Birchnat) showed higher median SOC stocks compared to all other tested landcover types for each of the tested intervals and the whole 30 cm (Abbildung 16). In comparison to the revegetated grassland sites, all afforested birch sites showed a lower SOC stock, which was most pronounced in the sampling intervals 10-20 and 20-30cm (Abbildung 16). And, the SOC stock of the grassland sites reached almost the level of the SOC stock of the remnant birch woodlands. After 50 years of vegetation growth, the grassland sites showed a considerable higher SOC stock (∆ 20 t ha^{-1}) than the un-treated, barren land after 50 years.

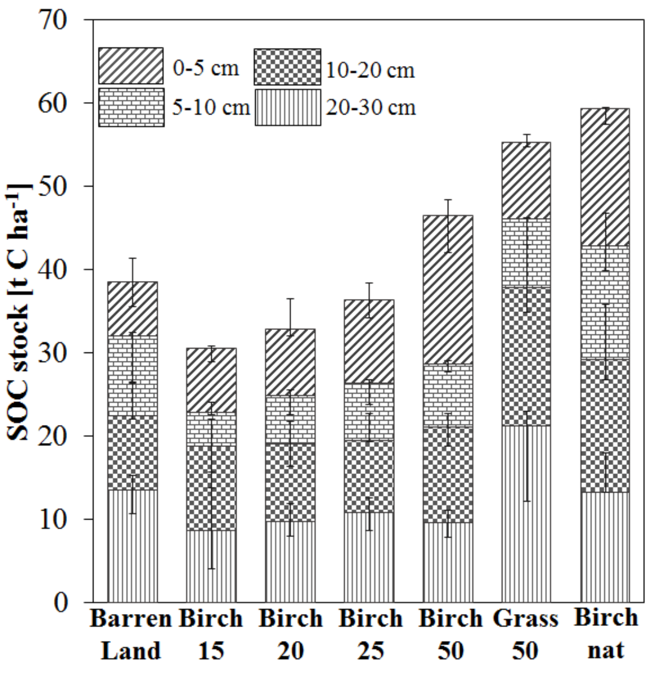

Abbildung 16: The median soil organic carbon stocks [t C ha^{-1}] in the mineral soil of the studied eroded (Barren Land), reclaimed (Grass50, Birch15, Birch20, Birch25 and Birch50) and old-growth (Birchnat) sites. The range of the error bars is showing the data range (minimum and maximum value). The different shadings indicate the four sampling depths (0-5cm: diagonal lines; 5-10cm: rectangular squares; 10-20cm: b,w squares; 20-30cm: vertical lines). Within a sampling depth, significant differences (Mann-Whitney U Test, $p \leq 0.05$) between the age classes are indicated by different letters. Further, significant differences (Mann-Whitney U Test, $p \leq 0.05$) between the total studied soil depth (0-30cm) are shown above the stacked columns.

4.3.3 The SOC in the analyzed fractions

During the development of birch woodlands, the SOC quality showed a continuous increase of the median mass of the POM material (>63 µm and <1.8 g cm^{-3}) in the top 30 cm of the soil, Barren Land: 5, Birch15: 43, Birch 20: 53, Birch 25: 51, Birch 50: 174 mg g^{-1} soil, respectively. In comparison, the median POM mass of Grass50 and Birchnat was about 24 and 95 mg g^{-1} soil, respectively. Thus, the supply of organic material as SOC source was higher at afforested birch stands compared to other tested sites and it increased exponentially with the age of the afforested stand. The different potentials of organic material input at the grassland system and the different old birch systems were also visible by comparing the POM mass found in the top 5 cm. It was two times higher at Birch15 (35 mg g^{-1} soil) than at Grass50 (17 mg g^{-1} soil) in 0-5 cm soil. In the same sampling layer, Birch50 stored about 155 mg g^{-1} POM material compared to 60 mg g^{-1} at Birchnat. In deeper sampling intervals, the same pattern of the POM mass was found. However, it occurred on distinct lower level. Hence, it is supposed that the SOC pools of all vegetated sites were mainly supplied by the aboveground living biomass of birch plants, the understorey vegetation and the dead organic litter pools, respectively, and less by the belowground biomass.

The SOC change was further analyzed by comparing the SOC concentrations of the fractionated soil samples. The SOC at Barren Land which represented the status before any reclamation activities started was mainly stored in the '< 63 µm' fraction (Abbildung 17) and the C concentration was distinctly higher than those of the afforested birch sites (Birch15-Birch50) in all depth intervals. At Grass50, the SOC was dominantly stored in the '< 63 µm' fraction. The SOC fractionation revealed further that the POM-C concentration increased along the afforested time span in all sampled intervals (Abbildung 17). Similar patterns were found for C concentration of the '< 63 µm' fraction in the top two depth intervals. Birch50 had a higher POM-C concentration in 0-5 cm but a lower concentration in deeper intervals compared to Birchnat. In the top 10 cm, the SOC which is stored in the HF fraction had also increased along the time span but it was time-shifted compared to the POM-C and '< 63 µm'-C. After fifty years of reclamation with fertilizer and grass seeding, Grass50 showed a trend to increase in all analyzed fractions compared to Barren Land within the top 5 cm. Due to the high variability in deeper sampling intervals, the SOC concentration did not show a trend between Barren Land and Grass50 (Abbildung 17). The DOC stock values indicated that the amount of dissolved organic carbon increases during the establishment of birch woodland and tends towards the concentration level of Birchnat. At the birch sites, the highest DOC concentrations were found in the top sampling interval and decreased with soil depth (Abbildung 17). At Barren Land and Grass50, the DOC concentration was higher in the subsoil than in the topsoil (Abbildung 17).

Further, in the top 5 cm, the SOC at the afforested sites was dominated by the POM fraction. However the '< 63 µm' fraction was responsible for the majority of the SOC at Barren Land and Grass50. It seemed that after 50 years of afforestation, the highest sequestration potential compared to Birchnat was in the HF fraction for the top 5 cm. In the 5-10 cm layer, SOC was mainly stored in the '< 63 µm' fraction at all tested sites. In deeper sampling intervals SOC pool in the '< 63 µm' fraction became even more dominant.

Over the whole 30 cm depth, the dominant fraction is the '< 63 µm' fraction for all studied landcover types except of Birch50 (Abbildung 18). At the afforested sites, the POM fraction became, however, more important and played a dominant role at Birch50. At Birchnat, a considerable high amount of carbon is further stored in the HF-fraction. The DOC concentration showed an increase along the sequence of the birch stands and was highest at Birchnat.

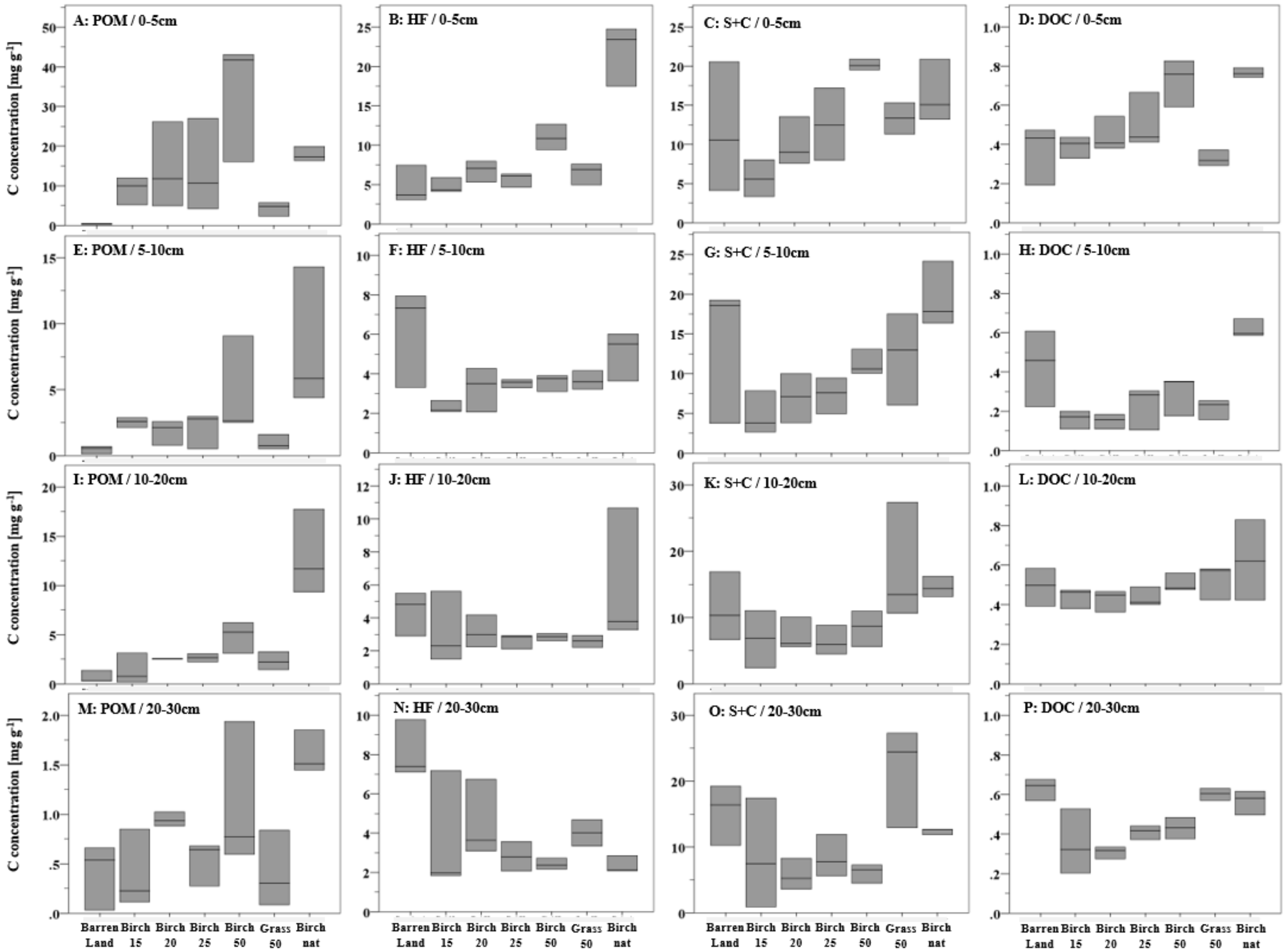

Abbildung 17: SOC concentration [mg g-1] of the fraction POM (A-M), HF (B-N), '< 63 µm' (C-O) and DOC (D-P) divided into the sampled soil depths (0-5, 5-10, 10-20 and 20-30 cm) for the studied reclaimed (Grass50, Birch15, Birch20, Birch25 and Birch50), eroded (Barren Land) and old-growth (Birchnat) sites. The boxes are showing the minimum, median and maximum values. Notice that the scale of the Y-axis is variable.

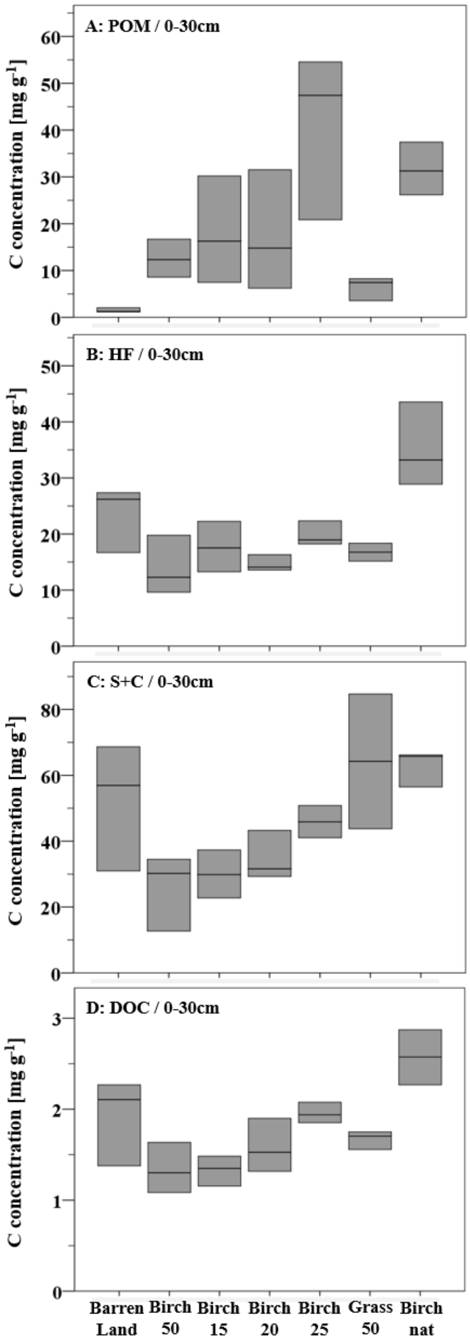

Abbildung 18 Cumulated carbon concentrations [mg g^{-1}] (0-30cm) within the analyzed SOC fractions for the studied reclaimed (Grass50, Birch15, Birch20, Birch25 and Birch50), eroded (Barren Land) and old-growth (Birchnat) sites. The boxes are showing the minimum, median and maximum values. Notice that the scale of the Y-axis is variable.

4.3.4. The fractionated SOC stocks

On eroded land, the bulk SOC stock consisted mostly of carbon which was stored in the '< 63 µm' (66 %) and HF (30 %) fractions, respectively (Tabelle 8). Even on grassland (Grass50), most of the carbon stock (94 %) was stored in these two fractions and only a marginal amount was found in the POM-fraction (3 t C ha^{-1}). Turning eroded land into birch woodland led to a continuous increase (+12 t C ha^{-1}) of the SOC stock in the POM fraction and resulted in a median POM-SOC stock which was higher at Birch50 than Birchnat. During 50 years of birch woodland establishment, the SOC stocks of the HF and '< 63 µm' fractions seemed to stagnate at about 9 t C ha^{-1}. The HF and '< 63 µm' stocks were reported on lower levels for Birch50 than for Birchnat. Further, old-growth birch woodlands (Birchnat) stored half of the total SOC stock in the '< 63 µm' fraction and equal proportionated in the POM and HF fraction (Tabelle 8).

Tabelle 8: The SOC stocks [t C ha^{-1}] for 0-30 cm explained by the analyzed SOC fractions. The median value and the minimum and maximum values (in paranthesis) are given.

Type	SOC stock			
	POM [t C ha^{-1}]	HF [t C ha^{-1}]	S+C [t C ha^{-1}]	DOC [10^1 t C ha^{-1}]
Barren Land	0.7 (0.2; 1.3)	11.5 (9.1; 13.0)	26.4 (14.2; 33.3)	2.0 (1.2; 2.7)
Birch15	4.9 (3.5; 7.6)	7.3 (5.5; 11.9)	17.4 (5.9; 25.3)	1.1 (0.6; 1.9)
Birch20	5.8 (3.6; 9.2)	9.9 (7.6; 11.9)	16.5 (12.3; 20.9)	1.1 (0.8; 2.1)
Birch25	6.3 (3.2; 9.3)	8.7 (7.1; 9.8)	19.9 (16.3; 26.0)	1.4 (1.1; 2.1)
Birch50	13.2 (7.1; 17.3)	9.0 (8.2; 9.8)	23.5 (18.3; 25.5)	2.5 (1.7; 3.1)
Grass50	3.1 (1.7; 4.0)	9.1 (8.0; 10.5)	41.5 (25.8; 53.9)	2.8 (1.7; 3.3)
Birchnat	11.5 (9.5; 16.2)	12.8 (10.3; 19.8)	32.4 (29.5; 36.8)	3.7 (3.1; 5.3)

4.3.5 The sequestration rate of SOC

The bulk SOC stock (0-30 cm) of Barren Land was used as reference SOC stock when the SOC sequestration rates were calculated. Revegetation (0.34 t C ha^{-1} yr^{-1}) on severely degraded land bound annually 2 times more than afforestation (0.16 t C ha^{-1} yr^{-1}) over a period of 50 years (Tabelle 9). Due to the higher SOC stocks at Barren Land than at the afforested birch sites (Abbildung 16), afforestation led to a release of soil organic carbon (between -0.53 and -0.09 t C ha^{-1} yr^{-1}) until the stand age of 25 years. Hence, it seems that afforestation leads to a loss of soil organic carbon during the first decades compared to Barren Land. The comparison between different birch stages however revealed an annual storage of organic carbon in the soil which was higher than 0.4 t C ha^{-1} yr^{-1}. The same calculations were done for the SOC stocks within each fraction (Tabelle 9). The annual SOC stock changes (0-30 cm) within each tested fraction revealed that the afforestation efforts showed an increase of the POM-C stock but no clear trends for the HF and the '< 63 µm' stocks (Tabelle 9).

The sequestration rates within the top 10 cm were calculated to educe the effect of the afforestation and hence the biomass input into the soil system. The median POM-mass as an indicator for the input of fresh biomass material into the soil showed a continuous increase during the establishment of mountain birch woodlands (Barren Land: 5; Birch15: 42; Birch20: 49; Birch25: 46; Birch50: 166 mg g^{-1} soil). According to this, the annual increase of the POM-mass was between 1.66 and 3.20 mg g^{-1} soil. The concentration of the POM-mass and the annual allocation at Grass50 were only 24 mg g^{-1} soil and 0.38 mg g^{-1} yr^{-1}. Compared to revegetation to grassland (0.03 t C ha^{-1} yr^{-1}), afforestation by birch revealed negative sequestration rates (Birch15: -0.28, Birch20: -0.11 t C ha^{-1} yr^{-1}) until the stand age of 20 yrs due to the higher SOC stocks at the eroded sites compared to the birch sites (Abbildung 16). Afterwards, an increase of the SOC accumulation (Birch25: 0.04, Birch50 0.19 t C ha^{-1} yr^{-1}) was observed until the age of 50 yrs. Between the different old afforested birch stands, the median annual increase of the SOC stock varied between 0.34 and 0.63 t C ha^{-1} (Tabelle 10).

The present study found a continuous increase of the POM-mass, the bulk SOC concentration and the SOC stock, respectively, with the increase of age of the afforested site (Tabelle 6 and Abbildung 16). Further, the median POM-mass showed also an increase (Birch15: 42; Birch20: 49; Birch25: 46; Birch50: 166 mg g^{-1} soil). Compared to the SOC stock of eroded land, it increased by 9 t C ha^{-1} (0.18 t C ha^{-1} yr^{-1}) during 50 yrs of birch growth. Further, the 50 yrs old afforested birch woodland showed a higher median POM-mass (Δ = 146 mg g^{-1} soil) and also a higher SOC stock (Δ = 8 t C ha^{-1}) than the restored grassland in the top 10 cm.

Tabelle 9: Estimated SOC sequestration rates [t C ha^{-1} yr^{-1}] for the SOC stock (Total SOC) and the SOC in the studied fraction (POM-C, HF-C and '< 63 µm'-C) for 0-30 cm. The values of the DOC fraction are positive and negative in terms of loss and sequestration, respectively. The rates are calculated for the time spans between Barren Land and the different old revegetated study sites and between the different old, afforested birch sites. The median value and the minimum and maximum values (in paranthesis) are given.

Time span	Rate of sequestration				Rate of loss
	Total SOC [t C ha^{-1} yr^{-1}]	POM-C [t C ha^{-1} yr^{-1}]	HF-C [t C ha^{-1} yr^{-1}]	'< 63 µm'-C [t C ha^{-1} yr^{-1}]	DOC [kg C ha^{-1} yr^{-1}]
Barren Land to Birch15	-0.50 (-0.53;-0.35)	0.28 (0.21; 0.42)	-0.24 (-0.28; -0.07)	-0.55 (-0.60; -0.53)	-5.7 (-6.2; -4.0)
Barren Land to Birch20	-0.27 (-0.06; -0.28)	0.26 (0.17; 0.39)	-0.07 (-0.08; -0.05)	-0.50 (-0.62; -0.09)	-4.6 (-5.1; -1.8)
Barren Land to Birch25	-0.09 (-0.14; 0.12)	0.22 (0.12; 0.32)	-0.11 (-0.13; -0.08)	-0.26 (-0.29; 0.08)	-2.3 (-2.5; -0.4)
Barren Land to Birch50	0.16 (0.09; 0.21)	0.25 (0.14; 0.32)	-0.05 (-0.06; -0.02)	-0.06 (-0.16; 0.08)	1.1 (0.7; 1.1)
Barren Land to Grass50	0.34 (0.26; 0.37)	0.05 (0.03; 0.05)	-0.05 (-0.05; -0.02)	0.30 (0.23; 0.41)	1.1 (1.0; 1.6)
Birch15 to Birch20	0.45 (-0.05; 1.27)	0.20 (0.03; 0.33)	0.43 (-0.01; 0.52)	-0.18 (-0.88; 1.28)	0.3 (-3.3; 4.7)
Birch20 to Birch25	0.70 (0.38; 0.81)	0.02 (-0.08; 0.08)	-0.24 (-0.42; -0.10)	0.80 (0.67; 1.03)	6.7 (5.4; 7.9)
Birch25 to Birch50	0.31 (0.30; 0.40)	0.28 (0.16; 0.32)	0.01 (0.00; 0.04)	0.08 (-0.02; 0.14)	4.0 (2.6; 4.4)

Tabelle 10: Estimated SOC sequestration rates [t C ha^{-1} yr^{-1}] for the SOC stock (Total SOC) and the SOC in the studied fraction (POM-C, HF-C and '< 63 µm'-C) for 0-10 cm. The values of the DOC fraction are positive and negative in terms of loss and sequestration, respectively. The rates are calculated for the time spans between Barren Land and the different old revegetated study sites and between the different old, afforested birch sites. The median value and the minimum and maximum values (in paranthesis) are given.

Time span	Rate of sequestration				Rate of loss
	Total SOC [t C ha^{-1} yr^{-1}]	POM-C [t C ha^{-1} yr^{-1}]	HF-C [t C ha^{-1} yr^{-1}]	'< 63 µm'-C [t C ha^{-1} yr^{-1}]	DOC [kg C ha^{-1} yr^{-1}]
Barren Land to Birch15	-0.28 (-0.40; 0.16)	0.28 (0.21; 0.33)	-0.08 (-0.04; -0.14)	-0.45 (-0.49; -0.07)	-2.6 (-4.3; 1.1)
Barren Land to Birch20	-0.06 (-0.11; 0.17)	0.20 (0.11; 0.34)	-0.03 (-0.00; -0.08)	-0.25 (-0.32; 0.04)	-1.5 (-1.6; 0.9)
Barren Land to Birch25	0.04 (0.01;0.20)	0.18 (0.08; 0.29)	-0.02 (0.00; -0.06)	-0.13 (-0.20; 0.09)	-0.1 (-0.4; 1.0)
Barren Land to Birch50	0.19 (0.17; 0.25)	0.22 (0.11; 0.27)	0.00 (-0.01; 0.01)	-0.03 (-0.07; 0.10)	1.7 (1.5; 1.7)
Barren Land to Grass50	0.03 (-0.02; 0.13)	0.04 (0.02; 0.04)	0.00 (-0.01; 0.00)	0.01 (-0.01; 0.06)	-0.5 (-1.0; 0.4)
Birch15 to Birch20	0.38 (0.18; 0.97)	-0.05 (-0.19; 0.36)	0.12 (0.11; 0.13)	0.36 (0.21; 0.49)	1.2 (0.4; 7.1)
Birch20 to Birch25	0.31 (0.27; 0.63)	0.09 (-0.03; 0.13)	0.03 (0.02; 0.04)	0.31 (0.27; 0.32)	4.2 (1.2; 5.5)
Birch25 to Birch50	0.33 (0.30; 0.34)	0.25 (0.13; 0.26)	0.02 (0.02; 0.04)	0.08 (0.07;0.10)	3.5 (2.1; 3.9)

4.4 Discussion

4.4.1 SOC stock changes during the ecosystem restoration

In Iceland, restoration activities on well-drained soils can be revegetation (e.g. Arnalds et al., 2013), afforestation on heathland (e.g. Ritter, 2007) or afforestation on severely degraded soils (present study). Besides these types of land-use and land cover change, natural succession of vegetation is another process which allocates more carbon in the vegetation and the soil (Vilmundardóttir et al., 2015). The initial SOC conditions at the starting point of restoration can differ due to different initial vegetation covers or soil characteristics like SOC concentration. This is relevant when SOC stock changes are compared.

Degraded soils generally contain a SOC stock between 1 and 45 t C ha^{-1} before any restoration activities start (Óskarsson et al., 2004; Arnalds et al., 2013). The present study calculated a median SOC stock of 40 t C ha^{-1} (Abbildung 16) for such soils and accompanies with the higher values given in the literature. This implies that the soils of Barren Land consists certain amount of SOC due to former soil formation processes and SOC accumulation which occurred before the soil profile was capped during soil erosion processes. On the other hand, the SOC stocks of Barren Land are significantly lower than in soils under well-established and not-degraded ecosystems. Hence, the present study confirmed the high sequestration potential of these severely degraded soils (Lal, 2009). The Brown Andosol at an un-disturbed birch woodland contained a median SOC stock of 59 t C ha^{-1} which is a lower than reported by Óskarsson et al. (2004) (68 t C ha^{-1}). In regard to the restoration of wooded ecosystems, the SOC stock of 59 t C ha^{-1} acted as target level.

Investigating the effect of afforestation concerning SOC sequestration, the present study found a continuous increase of the median SOC stock with the age of the stand (Birch15: 31; Birch20: 33; Birch25: 36; Birch50: 46 t C ha^{-1}) (Abbildung 16). Consequently, afforestation can increase the SOC stock in the soil. Contrary to the SOC stock increase in the present study, Ritter (2007) published a SOC stock of about 40 t C ha^{-1} (0-20 cm) for 26 and 97 years old birch stands and thus found no change after turning already vegetated heathland into birch woodland in Eastern Iceland. Snorrason et al. (2002) found a SOC stock (0-30 cm) of 65 t C ha^{-1} for a 54 yr old birch stand at Gunnarsholt which is a higher value than the SOC stock of Birch50.

The present study also examined restoration by revegetation and compared it with afforestation. Within 50 years, the revegetated sites which were restored by fertilizer and grass seeds showed an increase of the median SOC stock by 17 t C ha^{-1} (Abbildung 16). Within the present study, the 50 years old grassland sites showed a higher median SOC stock (+ 9 t C ha^{-1}) (0-30 cm) compared to the afforested birch stand with the same. By ignoring the carbon that is allocated in the biomass pool, the comparison indicates that soils sequester more organic carbon during revegetation than during afforestation with birch trees (Abbildung 16). Aradóttir et al. (2000) and Snorrason et al. (2002) also studied revegetated grassland sites near Gunnarsholt which showed a comparable site history than the grassland sites of the present study. Accordingly, Snorrason et al. (2002) reported a SOC stock (0-30 cm) of 54 t C ha^{-1} which was comparable with the SOC stocks of the present study. However, Aradóttir et al. (2000) found 28 t C ha^{-1} (0-20 cm) for a 46 years old grassland site compared to the median value of 34 t C ha^{-1} (0-20 cm) of the present grassland sites.

Natural plant succession and soil development on moraine till from glaciers is another typical process of landcover change in Iceland which can be also relevant regarding reducing countrywide carbon emissions (Vilmundardóttir et al., 2015). Vilmundardóttir et al. (2015) found a SOC accumulation which was recorded by SOC stocks between 0.9 and 13.5 t C ha^{-1} (0-20 cm) at sites with a maximum age of 120 years. The development of the SOC stock in soils of moraine till occurs on a distinctly lower level compared to the SOC stock which we found in degraded soils (median values between 23 and 36 t C ha^{-1}, 0-20 cm) with afforested birch vegetation with an age between 15 and 50 years (Abbildung 16). This different development can be explained by the natural and therefore extensive colonization of plants on fresh developing soils compared to the man induced enhancement of a more productive ecosystem on formerly developed soils which can still contain certain amounts of SOC. The comparison of the initial SOC stocks of these two studies shows further that the severely degraded soils of the present study must have sequestered organic carbon in former times compared to the SOC accumulation of glacial till which has been ice-free for 8 years (Vilmundardóttir et al., 2015). The birch woodland nearby the Skaftafell glacier showed a mean SOC stock of 49.5 t C ha^{-1} which is confirmed with the median value of the present study (Birchnat: 46 t C ha^{-1}; 0-20 cm) (Vilmundardóttir et al., 2015).

4.4.2 SOC fractionation enhance the understanding of the restoration success

In contrast to the expected vertical decrease of SOC (Jobbágy and Jackson, 2000), the sites of Barren Land showed higher SOC concentrations and stocks in the lower sampling intervals than in the topsoil (Tabelle 6 and Abbildung 16). The grassland and youngest birch stand showed the same characteristics (Tabelle 6). Arnalds and Kimble (2001) observed similar patterns for soils with lag-gravel surfaces, which developed through intense frost heave of coarse material and aeolian deposition. Strachan et al. (1998), Snorrason et al. (2002) and Kolka-Jónsson (2011) confirm this inverse vertical SOC pattern in disturbed and undisturbed soil pedons in the same region as the present study took part. Therefore, this inversion of the SOC stock with depth seems to be a common feature in south Iceland, which is the result of high volcanic, geomorphic and anthropogenic activities and disturbances (Kolka-Jónsson, 2011; Arnalds, 2015b)

A likely reason for this specific phenomenon could be the influence of water and wind erosion processes which affect well-developed Andosols with soil profiles up to 2 m in two different ways. According to Dugmore et al. (2009), soil erosion leads on one hand to a loss of fertile soil material and the development of soils remnants on erosional sites and on the other hand, aeolian sedimentation could lead to the thickening of soil profiles. The annual deposition rate is estimated to be more than 2.5 t ha^{-1} yr^{-1} for the study area (Arnalds, 2010). Following up on this theory, eroded soils can show a high spatial variability in thickness and might differ in their physical and chemical properties. This depends on the magnitude of erosion or deposition as well as on the specific pedon horizon of the topsoil which consist either of tephra, organic material or aeolian deposits (Arnalds, 2000; Arnalds et al., 2001). Consequently, this variability is most probably reflected in the calculated SOC stocks (Abbildung 16 or Bárcena et al., 2014), for example. Bárcena et al. (2014b) showed in their review that the variability is highest for afforested sites on former barren and degraded land in Iceland followed by former cropland, grassland or heathland. The same effect of soil disturbance in combination with the SOC stock development is conceptually showed by Rovira et al. (2015). Their suggestion is to apply a cumulative coordinates approach to correct any artifacts deriving from different sampling conditions (Rovira et al., 2015).

This high spatial variability is the reason why it is so challenging to find comparable test sites of the same parent material and, which only differ in the time since they were formed (Walker et al., 2010). Therefore, it is problematic to use a standardized, globally applied procedure (Aalde et al., 2006b) with fixed sampling depths and only total SOC stock measurements. The results give the impression that the conversion of eroded land into grassland is the more effective restoration type than afforestation (Abbildung 16) and further, soils (0-30 cm) of afforested birch trees seem to act as carbon sources until the age of 25 years (Tabelle 9). The landscape history in terms of erosional and depositional events however calls for a more complex interpretation of the data. The vertical fractionation of the SOC stocks clearly demonstrates that the eroded sites as well the grassland sites originally contained already higher SOC stocks than the ones found in the sampling intervals 5-10, 10-20 and 20-30 cm at the afforested birch stands (Abbildung 16). The physical SOC fractionation further revealed that the highest SOC concentrations (Abbildung 17) were found in the '< 63 µm' fraction at Barren Land and Grass50. And therefore, more than 70% of the SOC stocks (Tabelle 8) were found in the '< 63 µm' fraction at these sites. It is therefore more plausible that at Barren Land and Grass50 the SOC measured in deeper sampling intervals was sequestered in horizons during the soil formation in former times. Later, these C-rich horizons of the palaeosoils were buried by aeolian transported material and then again exposed by soil erosion. This assumption of sampling material of paeaosoils is underlined by the highest allophane and ferrihydrite contents at Barren Land and Grass50 (Tabelle 7) as a result of the weathering of soil minerals.

The separation of the SOC with a physical fractionation approach in combination with the splitting of the top 30 cm in four sampling intervals was able to distinguish between SOC pools of different origin which can be relevant for the determination of the reclamation success. Afforestation and the establishment vegetation on barren land evoked an increase of the SOC stock in the POM fraction. This increase was dependent of the vegetation age why more POM-C was found in deeper soil intervals at older afforested birch stands and at the natural growth birch woodland (Tabelle 8, Abbildung 18). On the other hand, the SOC fractionation further showed that substantial parts of the SOC stock are stored in the ‘< 63 µm’ fraction, especially in the soil of the severely degraded areas (Barren Land) and the re-vegetated grassland (Grass50) (Tabelle 8, Abbildung 18). Hence, the study clearly demonstrates that the SOC which is stored within the top 30 cm is mostly stored in the ‘< 63 µm’ fraction even in the severely degraded, barren soils (26 t C ha^{-1}) and only 12.5 t C ha^{-1} as increase of the POM-stock during 50 years can be attributed to the afforestation efforts.

4.4.3 Sequestration of carbon in the soil

Sequestration rates

The sequestration rates of soil organic carbon are commonly calculated with an initial SOC stock that represents the baseline regarding SOC accumulation at time for the soil depth of 30 cm (Aalde et al., 2006a). The present study used the SOC stock (0-30 cm) of Barren Land as reference SOC stock and calculated an annual carbon accumulation of about 0.34 t ha^{-1} for the conversion of barren land into grassland and 0.16 t ha^{-1} for the conversion of barren land into forest by afforestation within the period of 50 years. Further, until the age of 25 years birch stands showed even a release of carbon which was between 0.09 and 0.53 t C ha^{-1} yr^{-1}. As it was showed and discussed in previous chapters (Abbildung 16, chap. 4.4.2), the method (Aalde et al., 2006a) showed its disadvantages when the reclamation success on formerly severely degraded soils is monitored because of the relatively high SOC stock taking as baseline.

The fractionation of the SOC stocks in combination with the calculation of the sequestration rates revealed a steady input of organic material into the POM fraction during the establishment of birch vegetation (Tabelle 9). Interestingly, the annual changes of the POM stocks of the grassland sites are much lower than those at the old birch sites of the same age (0-30 cm). This is contrary to the relationship of living above- and below-ground biomass (root:shoot ratio) which is 3.70 and 0.23 in grasslands and deciduous forests (Jackson et al., 1996). We explain it by the hampered incorporation of aboveground litter due to denser surface vegetation cover and a lower in belowground biomass supply at the revegetated grassland sites. The stagnation of the sequestration rates (HF-fraction) or even the loss of carbon (‘< 63 µm’-fraction) is explained by the higher SOC stock in the mentioned fraction at Barren Land compared to the reclaimed birch sites (Tabelle 8). Reasons for the lower SOC stocks at the younger afforested birch sites compared to the barren sites can be related to the change to the SOC stock level due to changed amount of C-input and C-output (Guo and Gifford, 2002). This are: i) the formation of a litter layer at the birch sites which conserve parts of the aboveground dead biomass on top of the mineral soil and hence reduces the C-input to the soil (Guidi et al., 2014b), ii) the higher emission rates of CO_2 from soils during the establishment of birch vegetation compared to the rates of barren land which increases the C-output and iii) the loss of SOC by dissolved organic carbon (Abbildung 18). However, it seems to be most plausible that the SOC stock level was not equal at all tested soils before any vegetation growth, especially in the '< 63 µm'-fraction. It is therefore most crucial to select representative reference sites or to apply another study setup (e.g. repeated sampling) instead of using the chronosequence approach (Bárcena et al., 2014).

For the conversion into grassland, Arnalds et al. (2000) found higher bulk SOC accumulation rates (0.60 t C ha^{-1} yr^{-1}) for sites with an age of more than 50 years. This is considerably higher than found in the present study and calculated for 50 (0.06 t C ha^{-1} yr^{-1}) and 90 years (-0.01 t C ha^{-1} yr^{-1}) old restored grassland sites (Kolka-Jónsson, 2011). For afforestation with mountain birch, soils accumulate annually 0.47 within 65 years (Kolka-Jónsson, 2011). In the National Inventory Report (Hellsing et al., 2016), the country-specific removal factor is given as 0.51 t C ha^{-1} yr^{-1} for revegetation and afforestation activities and it is therefore assumed that soils of restored sites act as a carbon sink. The taken value of 0.51 t C ha^{-1} yr^{-1} is considerable higher than the presented annual changes (Tabelle 9). Based on our results, this indicates that the taken threshold seems to be too high for the SOC sequestration on severely degraded soils during restoration activities and further investigations based on a more robust dataset are needed.

The low or even negative SOC sequestration rates of the present study are mainly caused by the high SOC stock in the deeper sampling layers at the degraded and untreated sites (BarrenLand) (Abbildung 16) (explanations for this pattern see chap. 4.4.2). Hence, the present study also calculated the SOC sequestration rates for the top 10 cm and found that afforestation releases SOC during the first 20 years (< 0.28 t C ha^{-1}; median value) followed by a median annual SOC accumulation of 0.04 to 0.19 t C ha^{-1} between 25 and 50 years after turning severely degraded land into vegetated land with birch plant (Tabelle 10). Arnalds et al. (2013), reported for revegetation activities an annual SOC accumulation between 0.4 and 0.6 t C ha^{-1} (0-10 cm) during the first seven years of restoration on Vitrisols. During afforestation by Siberian larch on heathland in East Iceland, Ritter (2007) found an annual increase of 0.21 t C ha^{-1} (0-10 cm) in the mineral soil. After glacier retreat, the mineral soil (0-10 cm) accumulates between 0.07 and 0.11 t C ha^{-1} during the natural vegetation succession with mountain birch (Vilmundardóttir et al., 2015). In the case, natural vegetation, soil development, the initial stage of soil formation and the lacking of biomass and hence litter production are the reasons why the SOC accumulation occurs on a lower level compared to the other referred studies focusing on reclamation approaches on partly vegetated reference sites (heathland) and with the use of fertilizer (stimulation by N inputs, e.g. Gudmundsson et al., 2004). Nevertheless, the present study found comparable sequestration rates (0-10 cm) to those of Vilmundardóttir et al. (2015) and even lower rates for the soil depth of 30 cm by using the standardized technique of Aalde et al. (2006a).

Estimating the sequestration potential of severely degraded soil after afforestation by birch

Knowing the heterogeneous patterns of SOC stock at severely degraded soils, we still want to give a rough estimation for how long the soil within the top 30 cm of afforested birch stands can act as a C sink. For this, we only considered the sequestration rates (0-30 cm) between the different age classes of afforested birch stands (Tabelle 9) due to the reasons of the observed relatively high SOC stocks of the severely degraded soils (see above, Abbildung 16). The time weighted sequestration rate was calculated to be 0.38 t C ha^{-1} yr^{-1}. Based on this and the difference of the median SOC stocks between Birch15 (31 t C ha^{-1}) and Birchnat (59 t C ha^{-1}), the duration until the SOC stock of a natural grown birch woodland is reached takes 74 years. This time span is much shorter than the 250 years of possible SOC accumulation with 0.5 t C ha^{-1} published by Arnalds et al. (2013). The reason is that the severely degraded soils studied by Arnalds et al. (2013) contained about 4 to 5 t C ha^{-1} which is one-tenth of what was observed in the present study. An additional process has to be considered while the time of SOC sequestration is estimated. In the area where the study took place, the deposition rates of dust material are quite high (> 2.5 t ha^{-1} yr^{-1}) (Arnalds, 2010). This leads to a lower SOC concentration and hence lower SOC stocks (Arnalds, 2010). Due to the sedimentation and thickening of the soil, we, therefore, assume that the potential time, in which the soil can act as a sink, is expanded until the SOC stock level of undisturbed soils is reached.

Besides the increase of the carbon pool in the living biomass (Bjarnadottir, 2009; Hunziker et al., 2014), the results of the present study confirm that afforestation by the native tree species *Betula pubescens* Ehrh. *ssp. czerepanovii* is a successful strategy to enrich the soil organic carbon pool (Sigurðardottir, 2000; Snorrason et al., 2002). Nevertheless, the study also brings forward that up to 44 % of the SOC in the top 10 cm (Tabelle 8) is stored in an unprotected form (POM) and is easily accessible for the decomposition by faunal consumers. The physical fractionation of the soil further revealed that the SOC quality changes due to the different stages of the forest development (Tabelle 8, Abbildung 18). Compared to old growth birch sites, which have already passed the self-thinning stage, all afforested sites of the present study will experience a reduction of their biomass productivity, due to the forthcoming self-thinning process. This means that due to the diminished supply by litter material after 50 years, the quantity and quality of the SOC will change. It can be assumed that the ratio of the more labile POM-C stock will be reduced (Birch50: 0.42; Birchnat: 0.24) which accompanies with the decline of the bulk SOC lability. Another cause for the reduction of the ratio could be the incorporation into more stable organo-mineral complexes of the HF and '< 63 µm' fraction during decomposition. This could be the reason for the higher SOC concentrations and SOC stocks in these two fractions at Birchnat compared to Birch50 (Tabelle 8, Abbildung 18). And another part of the POM-C can be released to the atmosphere by passing the chemoorgano-heterotrophic metabolism of decomposer community.

4.4.4 SOC stabilization by volcanic clay minerals

In volcanic parent material, rapid crystallization of Al and Si and Fe results in the building of poorly crystalline (short-range order) morphological minerals called allophane, imogolite and ferrihydrite which can build hollow spherules, thread-like and tubular structures or well-aggregated spherical particles (McDaniel et al., 2012; Arnalds, 2015d). These minerals may play a key role in stabilizing soil organic carbon in volcanic soils due to their amorphism, high degree of hydration, the extensive specific surface area (200-1500 m^2 g^{-1}), the pH-dependent charge and the high reactivity (Torn et al., 1997; Basile-Doelsch et al., 2007; McDaniel et al., 2012; Arnalds, 2015d). The major stabilization mechanisms are either the formation of allophane- or ferrihydrite- humus complexes which is favored at pH > 5.0 or the building of metal-humus complexes which is more effective at low pH values lower than 5.0 (Arnalds, 2015d). In order to discuss the SOC stabilization with mineral clays in more detail, we additionally considered the SOC concentration of the '< 63 µm' fraction which represents also the fraction of the mineral clays (Abbildung 19; C, F).

The present study confirms the positive ($r = 0.66$) and negative ($r = -0.77$) correlations between the allophane concentration respectively the (Al_{pyr}+Fe_{pyr}):(Al_{ox}+Fe_{ox}) ratio and the pH value (Abbildung 19; A, D). Both correlations indicate a possible influence of the degree of vegetation cover on the amounts of allophane and the ratio Al_{pyr}:Fe_{pyr}, respectively. The allophane concentrations are highest in the soils which were un-vegetated (Barren Land). And on the other hand, the concentrations of Al and Fe bound to metal-humus complexes were highest in the top sampling intervals of sites with the longest vegetation covers (Birchnat, Birch50 and Birch25; dotted circle). We assume that the organic matter derived from the vegetation reduces the pH value and thus enhance the formation of metal-humus complexes (Arnalds, 2008, 2015f). According to the results of the present study, it seems that the SOC is preferably bounded to metal-humus complexes than to allophane clays (Abbildung 19; B, C, E, F).

The scatterplots comparing the allophane concentration with the bulk SOC concentration as well as the '< 63 µm' SOC concentration show no clear trends because most of the samples contained less than < 4 % of bulk SOC or '< 63 µm' SOC. The highest SOC concentrations were found in the upper sampling intervals at Birch25, Birch50 and Birchnat (dotted cyrcles). However, the allophane content is lowest in these cases (dotted cyrcle) which may be attributed to the fact that soil weathering and the formation of clay minerals takes longer than the allocation of soil organic carbon during birch growth. In regard to SOC sequestration during the reclamation of severely degraded land and soils, soil material of eroded and capped soil profiles passed most likely already through weathering processes and containing high amount of clay minerals. Hence, the carbon sequestration potential of these eroded soils may be relatively high (Lal, 2009). And, the fresh soil organic carbon originating from reclamation activities can be stabilized by the already existing clay minerals like allophane (Tabelle 7, Abbildung 19).

The stabilization of the SOC in form of metal-humus complexes seems to be undetermined (Abbildung 19; E, F). As above already mentioned, the upper most sampling intervals of the intense vegetated sites (dotted circle) laid decoupled from the nested scatters. For the soil intervals of these cyrcled sites, the formation of metal-humus complexes might be reasonable stabilization process of the SOC in the '< 63 µm' fraction due to its positive relationship (Abbildung 19; F).

4.5 Conclusion

The study aims to evaluate the SOC sequestration potential of afforestation on severely degraded soil in south Iceland. For this, we measured the SOC stocks of different old afforested birch stands and compared them with those of eroded and degraded soils, re-vegetated grasslands and woodlands which have escaped the soil erosion, respectively. In addition, the SOC quality of all sites was analyzed by soil fractionation. Based on these results, the study was able to give detailed evidence concerning the carbon sequestration potential of soil reclaimed by afforestation.

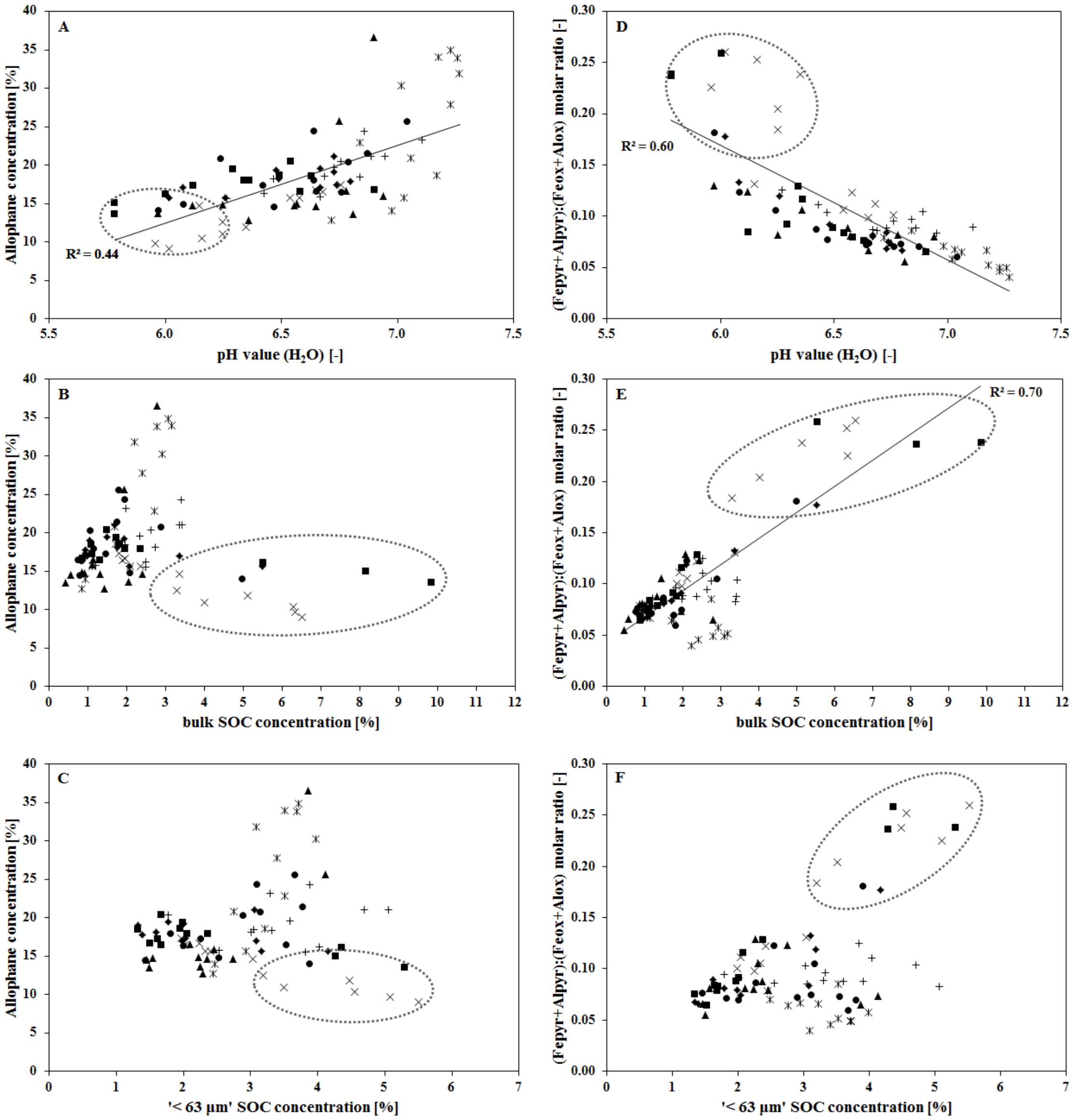

Abbildung 19 Relationship between common properties of volcanic soils. The charts are showing the allophane concentration [%] as a function of pH value (H_2O) [-] (A), bulk SOC concentration [%] (B) as well as of '< 63 µm' SOC concentration [%] (C) and the amount of Al and Fe in the form of organo-mineral complexes ((Fe_{pyr}+Al_{pyr}):(Fe_{ox}+Al_{ox}) molar ratio [-]) as a function of pH value (H_2O) [-] (D), bulk SOC concentration [%] (E) as well as of '< 63 µm' SOC concentration [%] (F). The observations (N = 84) are labelled based on the vegetation types: Barren Land (⚹), Birch15 (▲), Birch20 (●), Birch25 (◆), Birch50 (■), Grass50 (+) and Birchnat (×). The dotted circles are showing all samples of Birchnat (0-5 cm, 5-10 cm), all samples of Birch50 (0-5 cm) and one sample of Birch25 (0-5 cm).

The afforestation with mountain birch trees leads to a continuous increase of the SOC stock from 15 to 50 years old stands. The study found a maximum median increment of 8 t C ha^{-1} after 50 years of birch growth. Afforested birch stands can still accumulate SOC after 50 years of plant growth due to the lower SOC stock (Δ = 13 t C ha^{-1}) compared to naturally, old grown birch woodlands. However, due to the lower SOC stock at younger birch stands compared to the degraded sites, the sequestration rates (0-30 cm) are ranging between -0.53 and 0.16 t C ha^{-1} yr^{-1} and are therefore considerably smaller than reported by other studies or used in the National Inventory Report (0.51 t C ha^{-1}). By using the presented sequestration rates and ignoring the on-going sedimentation of dust material, it can be assumed that 85 years after starting restoration of degraded land with birch trees, the SOC stock of natural-growth birch woodland is reached.

The analysis of the carbon quality showed, independently of the site condition, more than 50% of the SOC is stored in the '< 63 µm' fraction. Further, a significant increase of the POM pool indicates the success of restoration. On the other hand, due to the increased amount of POM material, the SOC pool is more vulnerable to release C to the atmosphere by passing the heterotrophic metabolism.

The results of the present study also clearly shows that undertaking research on soil organic carbon patterns on severely degraded soils is of challenging character. With the applied physical fractionation technique, the present study was able to differentiate the SOC and therefore to evaluate the success of afforestation by mountain birch on a landscape with highly diverse soil patterns and SOC distribution. But due to this heterogeneity, the applied space-for-time substitution approach showed limitations because the different old study sites differ not only in the age of the vegetation cover but also in the characteristics of the parent material. In such cases, it would be more useful to use permanent plots and a long-term monitoring approach to assess the development of the soil during vegetation restoration, like it was performed by Arnalds et al. (2013) and Thorsson (in prep.) in Iceland, studied more general by Bárcena et al. (2014a) and suggested by Johnson and Miyanishi (2008). Another finding of the study is that the standardized soil sampling depth of 30 cm turns out to be a critical issue to evaluate the success of restoration regarding SOC sequestration on severely degraded soils. Thus, it needs to be applied with caution.

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KAPITEL 5

Shrubification in southwest Greenland: Estimating the influence on SOC by comparing birch vegetation with shrub-free vegetation within the tundra-boreal ecotone

Blick vom auslaufenden Hang des Narsaarsup Qaava Plateau nördlich von Igaliku Richtung Südsüdost über den Einarsfjord zu den Kargletschern im Bereich von 60.78°N/45.11°W. Aufgenommen von M. Hunziker am 16. Juli 2013.

Shrubification in southwest Greenland: Estimating the influence on SOC by comparing birch vegetation with shrub-free vegetation within the tundra-boreal ecotone

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Status: Das wissenschaftliche Manuskript ist in einer ersten Fassung von Dr. Chatrina Caviezel (Mitautorin) bis am 16. Mai 2017 kommentiert worden. In der Zwischenzeit hat die intensive Überarbeitung des Inhaltes stattgefunden.

Abstract

The recent and future raise in temperature results in an elevational and latitudinal shift of vegetation. Southern Greenland lies at the border of the boreal-tundra ecotone which is also part of the mountain birch zone. Thus, an expansion of the woody flora into tundra ecotones in the course of climate change can be expected. This "shrubification" influences the soil organic carbon pool. Hence, the aim of the study is to quantify and characterize the soil organic carbon by physical soil fractionation techniques under tall-shrub vegetation (mountain birch) and shrub-free vegetation in the boreal-tundra ecotone in southwest Greenland in order to give an estimation of the soil carbon sequestration potential on landscape level. Besides the SOC properties, the shrub and shrub-free sites were described according to ecologic variables (bedrock material, relief, microclimate, vegetation and soil properties) to explain any differences. During field work, the catena approach was applied. The results revealed that the SOC stocks (0-30 cm) generally varied between 54 and 148 t C ha^{-1}. Differences between the bulk SOC stocks of shrub and shrub-free vegetation were marginal. However, the POM-stock was considerably higher under shrub vegetation than under shrub-free vegetation while the SOC stock in the silt and clay fraction was higher under shrub-free vegetation. Hence, the study concludes that shrubification can increase the vulnerability of the SOC concerning its stability. There is an assemblage of interacting factors which controls the abundance and growth patterns of plant species. On catchment scale, the major controlling factors seem to be the bedrock material and the relief which regulates the local temperature regime and the quality of the substrate (coarse fraction, macronutrient concentrations). Regional scaled climatic conditions are not the only driver for the abundance and growth properties of bushes. Hence, the present study suggests implementing lithologic and topographic variables based on DEM into climate-vegetation models for more reliable estimations concerning the spatial distribution of shrub expansion and further the quantification of carbon pools and fluxes in the future.

Keywords: shrubification, climate warming, land cover change, landscape system, soil organic carbon stock, soil fractionation

5.1 Introduction

5.1.1 Environmental change at the boreal-tundra interface

Soils cover approximately 121 x 10^6 km^2 of the terrestrial surface and are the largest terrestrial carbon reservoir. Between 3200 and 4100 Pg soil organic carbon (SOC) are stored within the top 3 meters (Jobbágy and Jackson, 2000; Davidson and Janssens, 2006; Settele et al., 2014). Hence, soils contain three times as much carbon than Earth's atmosphere (Köchy et al., 2015). For the circumpolar region (17.8 x 10^6 km^2), it is assumed that the storage of SOC in the top 0.3 m and 3 m is about 217 and 1035 Pg SOC, respectively (Hugelius et al., 2014). Apparently, soils of the circumpolar region contain proportionally the largest SOC pool on a global scale.

Climate warming in the course of climate change has been two to three times higher in subarctic and arctic regions than measured as global average since 1960 (Serreze et al., 2000; Hinzman et al., 2005; Anisimov et al., 2007). The increase of temperature particularly affects the terrestrial ecosystems as for example the change in vegetation composition (Callaghan et al., 2004). An indicator for changes in the photosynthetic activity and thus for the “greening” within these regions is the assessment of changes in the Normalized Difference Vegetation Index (NDVI). Xu et al. (2013) and Zhu et al. (2016) found that one third of the Pan-Arctic has greened between the years 1982 and 2012. According to these studies, the greening was thereby mainly related to the increase of the summer warmth index by 5 °C. Beside the expansion of the graminoid-like species of the tundra biome into the polar deserts (Bhatt et al., 2010; Jeong et al., 2012), the higher NDVI values can be explained by the ingrowth of woody flora of the boreal biome into the tundra biome (Blok et al., 2011; Boelman et al., 2011). Models predict that between 11 and 55 % of the tundra biome will be replaced by woody vegetation until 2100 (Callaghan et al., 2004; Zhang et al., 2013; Pearson et al., 2013; Larsen et al., 2014). An increase of woody vegetation was also detected by several field (Sturm et al., 2001; Kullman, 2002; Jia et al., 2003; Tømmervik et al., 2004; Tape et al., 2006; Wilmking et al., 2006; Rundqvist et al., 2011; Hedenås et al., 2011; Jørgensen et al., 2015).

In the following, the article focuses on shrubs due to the scope of it. According to Payette et al. (2002) and Myers et al. (2011), expansion and densification of woody vegetation in the tundra biome is defined as the increase of shrub biomass by i) the change of growth form like the increase of the canopy height of the shrub cover, ii) the increase of the shrub cover by lateral growth of existing shrubs and recruitment between existing stands and iii) the latitudinal and elevational shift of the shrubline beyond the previous range limit. Further on, “shrub-tundra” can be sub-classified mainly by the coverage ratio and growth height as “low- and high-shrub tundra”, “erect dwarf-shrub tundra” and “prostrate dwarf-shrub tundra” (Kaplan et al., 2003; Myers-Smith et al., 2011). This succession of shrubby tundra vegetation types builds a transition zone with zonal patterns and different borderlines in elevational and latitudinal direction which consequently do not form a sharp bushline (Walker, 2000; Payette et al., 2002; Myers-Smith et al., 2011).

Arctic and sub-arctic regions of the northern hemisphere may be of particular concern considering the potential feedback of abiotic and biotic factors like energy fluxes, regional climate and soil-atmosphere exchanges of ecosystems on global warming (Myers-Smith et al., 2011; Elmendorf et al., 2012). These feedbacks seem to change most rapidly between the tall-shrub tundra and the dwarf-shrub tundra (Myers-Smith et al., 2011). Due to the highly sensitive character of SOC in regard to environmental changes (Batjes, 1996; Hoffmann, 2012; Pries et al., 2017), it is unclear how the large and vulnerable SOC reservoir will react relating to the highly pronounced climate warming and the predicted vegetation change in the future (Schuur et al., 2015).

5.1.2 Predicted vegetation change in southwest Greenland

The southwest of Greenland is part of the low-arctic to sub-arctic region and is climatically graduated as hyperoceanic, oceanic, suboceanic and subcontinental while the continentality increases along the fjords from the coastline towards the ice sheet (Feilberg, 1984). The vegetation shows the above mentioned zonal patterns and changes along fjords from “low arctic-willow shrub” towards “sub-arctic – willow shrub and birch shrub” (Feilberg, 1984). Sub-arctic birch forests were found in the protected areas of the inner fjords in South Greenland during several field surveys (Polunin, 1938; Böcher, 1979; Jacobsen, 1987)., Coniferous species only exist in arboretums of a total area of 150 ha (Nielsen et al., 2011). Hence, the vegetation composition of the most southern tip of Greenland can be described as the shrub-tundra ecotone which contains graminoids, dwarf-shrubs and tall-shrubs (Feilberg, 1984). The shift towards a woody vegetation has however different facets. The interface between the boreal and tundra biomes is built either by coniferous (spruce, larch, pine) or deciduous (birch, willow, poplar, alder) species and hence called forest-tundra or shrub-tundra (Payette et al., 2002; Miller and Smith, 2012).

The meteorological weather station in Narsarsuaq has recorded surface air temperature since 1873. The mean July temperature was 10.3 °C (1961-1999) which is attended by definition for subarctic conditions (10 °C isotherm during the warmest month) (Feilberg, 1984; Hanna and Cappelen, 2002). The analysis of long-term temperature data revealed an increase of the summer (+1.8 °C) and the winter (+4.4 °C) temperatures for the period 2001-2012 compared to the reference period 1881-1910 (Hanna et al., 2012). For the future, climate models predict an additional temperature increase of 3.3 °C ± 1.3 °C for southern of Greenland until the end of the 21th century (Masson-Delmotte et al., 2012).

Alongside with higher temperature, the Danish Meteorological Institute (DMI) modeled an increase of the growing season of approximately 2 months until 2100 (Christensen et al., 2016). The increase of temperature and the expansion of growing season, favor a land-cover change from the graminoid tundra vegetation to a shrub tundra also on less protected areas in Greenland (Normand et al., 2013; Westergaard-Nielsen et al., 2015). This leads to an increase of the biomass pool and hence also an allocation of the carbon in the biomass (Westergaard-Nielsen et al., 2015). However, it is unclear how the vegetation change affect the soil organic carbon due due to the strong linkage between biomass productivity and soil organic carbon (Guo and Gifford, 2002; Vesterdal et al., 2013). Further, available data concerning Greenlandic SOC stocks and its changes in relation to vegetation change (Nielsen et al., 2011) is missing. An another issue is that soil organic matter components are sequestered in functional pools (Sollins et al., 1996; von Lützow et al., 2008). Hence, the composition of the bulk SOC can change during shrubification and the accompanying change of C-input.

In order to estimate the effect of shrubification of the shrub-free tundra on the soil organic carbon, the present study quantifies the SOC stocks and characterizes the SOC patterns of shrub and shrub-free vegetation on landscape level. The study considered the different settings of the landscape like geologic and topographic patterns, relief, geomorphic forms and vegetation forms to follow a more holistic approach. Therefore, additional site parameters were recorded which are relevant for the abundance and growth of plants and hence influence the SOC indirectly. We suppose that the highest SOC stocks are found at well-established birch stands with the highest ratio of un-protected organic material due to the highest productivity. Hence, the increase of the birch area would lead to an increase of the SOC reservoir in the landscape in the future as it was supposed by Tømmervik et al. (2009).

5.2 Methods

5.2.1 Study area

The study took place in the surrounding of Igaliku (60° 59′ 27″ N, 45° 25′ 9″ W) between the head of Igalikup Kangerlua (Einar’s Fjord) and Tunulliarfik fjord (Erik’s Fjord) (Abbildung 21). The study area stretches from the “West” plateau over the south-facing “Valley” to the “East” plateau. The plateaus reach elevations between 260 and 280 m asl. in the west and 220 to 280 m asl. in the east, respectively. The valley stretches over about 2000 m in the southern part and narrows towards north. The bottom of the valley lies at 50 m asl. and its slopes reach elevations of 250 m asl. in the northern part (Abbildung 21).

Geologically, the study area lies within the Ketilidian basement, which was formed 1850–1725 Ma ago (Brooks, 2012). It consists of granitic rocks of the Julianehåb batholith, which are partly overlain by Gardar sandstone sediments and lavas with outcrops of basaltic magmas and the intrusion of early dyke swarms called Gardar intrusions (Brooks, 2012). The bedrock of the “West” plateau is dominated by sandstones of the Eriksfjord Formation. In the “West”, the landscape is rippled due to the glacial-shaped upcoming bedrock. In general, the summits and slope shoulders are substrate-free and hence without any vegetation growth. Due to the inexistent infiltration of water, the depression zones act as traps for sediments and water. Thus, vegetation growth is concentrated to these areas where ponds and swamps dominate (Abbildung 22). The bedrock and also the parent material of the soil on the “East” plateau originates from the Igaliko Complex of the Gardar intrusions which mainly consists of nepheline syenite and its weathered derivatives (Sørensen et al., 2006). Due to the weathered and porous bedrock material, only few surface water courses exist. Active gullies as erosion forms that indicate sub-surface water flows within loose substrate material were found. As it shown in Abbildung 22, the sub-area “East” showed a low vegetation cover which mostly consists of graminoid and prostrate dwarf-shrub vegetation. The bedrock of the “Valley” mainly consists of Julianehåb Granite of the Ketilidian basement while the parent material of the soils in the valley bottom and at its hillslopes however are sediments of fluvio-glacial transported plutonic rock material (nepheline syenite), fluvial transported sandstone sediments as well as gravitational moved stones and boulders. The slopes of the “Valley” are well-vegetated by established willow, birch and grassy vegetation. The plain of the valley is composed of a mostly un-vegetation fluvial fan and a valley bottom dominated by ground moraine material with dwarf shrub and graminoid vegetation (Abbildung 22).

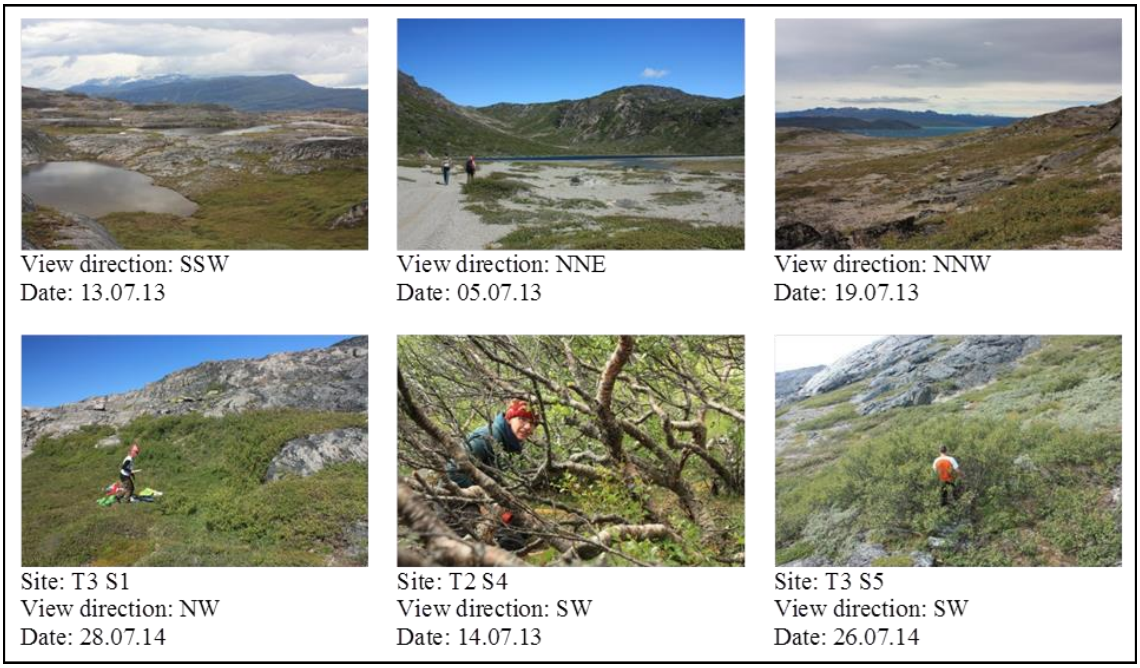


Abbildung 20: Impressions of the three sub-areas (top) and selected birch test sites (bottom) at “West” (left), “Valley” (middle) and “East” (right). Lithologic and pedologic characteristics of the three sub-areas are given in the text.

The meteorological weather station in Narsarsuaq provides the climatic conditions for the study area. The mean annual precipitation (1961-90) is 615 mm. The mean annual temperature lies at 0.9 °C, with July means of 10.3 °C and January means of -6.8 °C, respectively (Cappelen et al., 2001).

After Feilberg (1984), the vegetation in the area is characterized by local willow (*Salix glauca*) and mountain birch (*Betula pubescens* Ehrh.) copses on lower elevation and in sheltered sites. On less protected areas dwarf-shrub willow heath (*Salix glauca*) and understorey of herbs as clustered Lady's mantle (*Alchemilla glomerulans*) can be found. Above the elevational limit of willow heath, depending more on microclimatic and soil conditions than on defined elevation, vegetation is characterized by wind exposed fjell fields, where vegetation is mainly open. Species include dispersed and low growing patches of lichens, dwarf shrubs like Arctic willow (*Salix arctica)* and Common juniper (*Juniperus communis),* and grasses, among other Wood-rush (*Luzula confusa)* and Moss campion (*Silene acaulis)* (Feilberg, 1984; Madsen, 2014). Lichen dominated heath type is mostly found at higher elevations (Madsen, 2014). Further on, grassland slopes dominated by different species of natural grasses (*Anthoxanthum odoratum*, *Deschampsia flexuosa* and subordinatied *Alchemilla alpina*), moist fens with sedges and peat mosses (*Carex rariflora*, *E. scheuchzeri*, *Sphagnum*) (Feilberg, 1984; Madsen, 2014) can be found.

On small-scale, the soil atlas of the northern circumpolar region defines the dominant soil type in the inner parts of the southwestern fjords of Greenland as mollic Umbrisol which stands for soils with a dark, acidic surface horizon that is rich in organic matter (FAO, 2006b; European Commission, 2010). More detailed information concerning the pedogenesis is sparse. Distinct studies on the subarctic soils in South Greenland are mostly limited to archaeological research (Jacobsen, 1987; Rutherford, 1995; Adderley and Simpson, 2006; Edwards et al., 2011). Soils of former Norse settlements are described as generally well-developed and sandy with high amount of coarse fraction (Rutherford, 1995; Adderley and Simpson, 2006). Rutherford (1995) mentioned a high organic content in surface horizons as decomposition takes place slowly, but described also profiles showing high amount of carbon in subsurface horizons. Further on, soils are characterized acidic with pH around 4 (0.01 M $CaCl_2$) (Rutherford, 1995). Podzolization as soil forming process due to the sandy texture of the soils and the humid climate condition was observed (Jacobsen, 1987). Rutherford (1995) and Jacobsen (1987) conclude that the investigated soils around the former Norse settlements in South Greenland were developing through the entire Holocene. According to Feilberg (1984), Brown earth and Podzols occur in densely vegetated areas, while shallow Syrosems and Ranker developed where vegetation is scattered or constitutes just a continuous cover.

5.2.2 Data collection and analysis

Soil sampling and laboratory analysis

Field data was collected during the two field campaigns in summer 2013 and 2014. According to the geo-ecological settings (chapter 5.2.1), the study area was divided into three major sub-areas (Abbildung 21). The 22 study sites were placed semi-randomly along four transects. The catena approach was applied to represent the distribution of the environmental characteristics within the three sub-areas. In order to characterize the SOC patterns of birch woodlands and bush-free vegetation, the study sites contained 13 birch and 9 bush-free sites. Each site was characterized by its position in the landscape according to given topographic categories (FAO, 2006b). The soil and vegetation per site was examined more intensive within an area of 10 m by 10 m.

In Greenlandic soils, soil organic carbon can be found at least until 100 cm depth (Henkner et al., 2016; Petrenko et al., 2016). In order to compare SOC patterns of birch stands at different landscape elements, the sampling depth was however set to 30 cm because it corresponds the common depth interval for SOC stock inventories according to the IPCC guidelines for national greenhouse gas inventories (Aalde et al., 2006b). A second argument for the use of the mentioned sampling depth is that the top 30 cm contains most of the belowground living root biomass at grassland and birch areas, respectively, and thus, comprises the belowground organic carbon sources (Snorrason et al., 2002; Bjarnadottir et al., 2007; Hunziker et al., 2014). At each birch site, five soil pits were randomly placed within one half of the crown diameter of a dominant mountain birch (*Betula pubescens* Ehrh.) shrub within the 10 m by 10 m plot. The mineral soil was sampled after removing the organic layer. Per pit, the soil was sampled with a cylindric metal core (Eijkelkamp Soil & Water, Giesbeek) of 100 cm^3 volume and 5 cm in diameter at given soil intervals (0-5, 5-10, 10-20 and 20-30 cm). Due to the focus on SOC differences between different landscape elements respectively land-cover types, the five sub-samples were mixed in order to form one composite sample per site to make the dataset more robust. In total, 52 composite samples were collected.

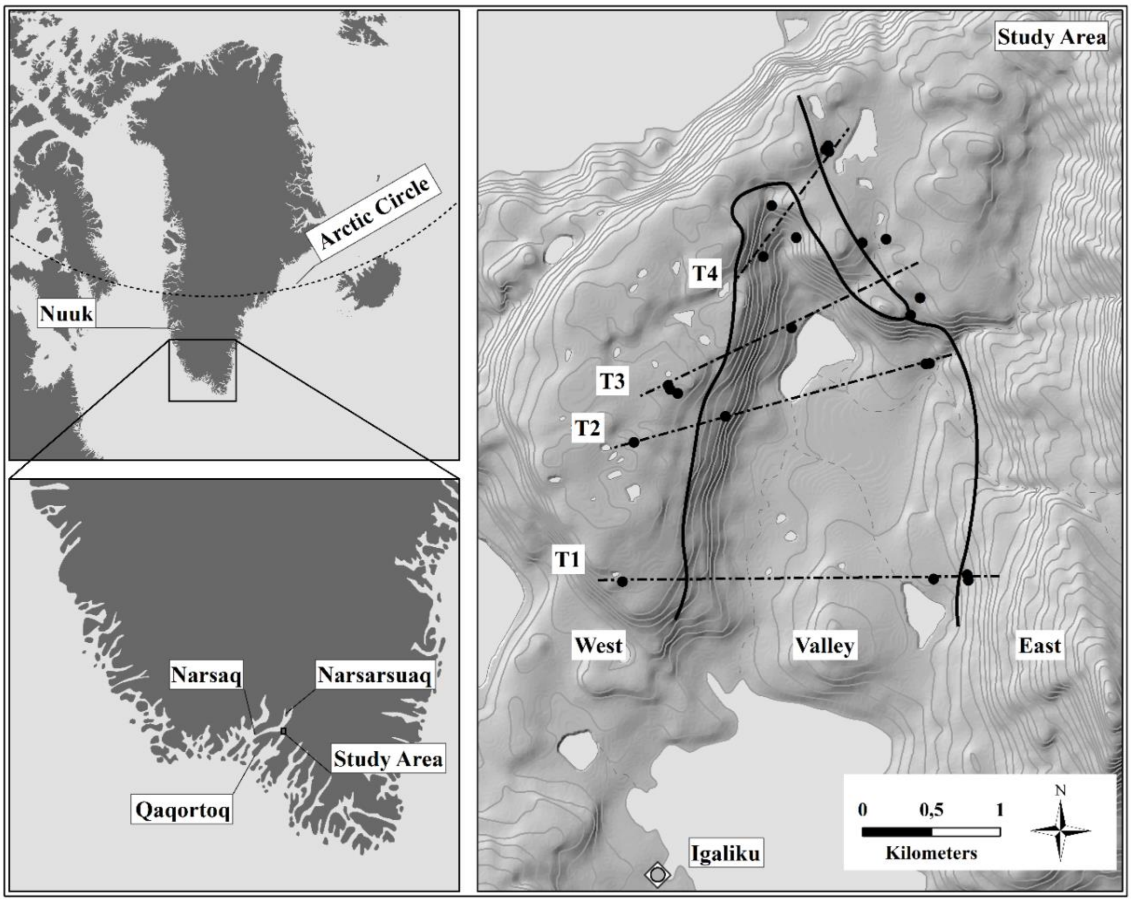


Abbildung 21: The field survey took place North of Igaliku in the inner fjords of southwestern Greenland. The black lines indicate the boundaries of the three sub-areas (West; Valley; East) representing different landscape units based on geologic, topographic and geomorphic patterns. The study sites (black dots) were chosen semi-randomly along 4 transects (dashed lines). Further, elevation lines (grey lines), water courses (dashed grey lines) and lakes, fjords as well as ponds (grey) are shown. The spatial data is displayed in the projection "UTM WGS 84 23N", in each case.

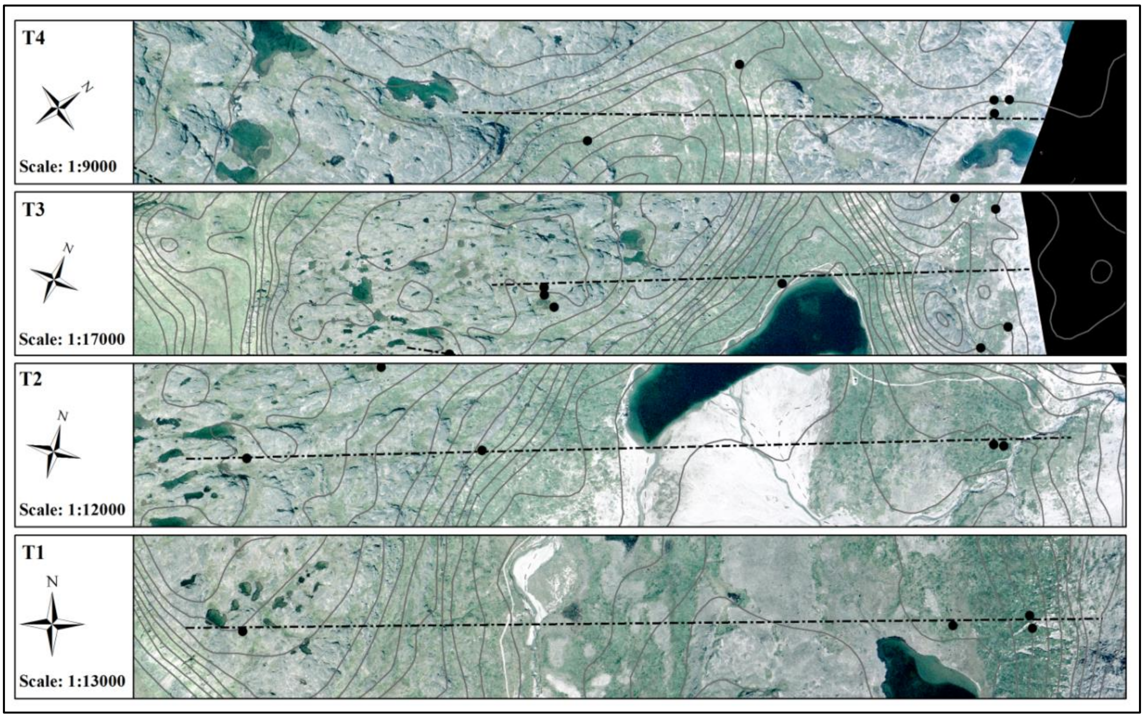


Abbildung 22: More detailed view of study sites (black dots) and the different landscape elements along the four transects (T1-T4) with the areal photograph from 2006 (© GEUS) in the background.

On the shrub free sampling sites, soil sampling techniques were adapted for the sampling in 2014. Due to the different sampling depths, depth of given results will be indicated in the following. In 2013, the sampling was performed with the cylindric metal core at the soil intervals 0-5 and 5-10 cm. In 2014, the soil within the 10 m by 10 m test site was randomly sampled by a hand auger reaching rooting depths of 30 cm. The composite sample of these sites consisted of 15-25 auger samples. The dataset consisted of 17 soil samples taken at the shrub-free sites. At the sites which were sampled by hand auger 3 soil samples were taken by a metal cylindric core of 100 cm^3 volume and 5 cm length at 12-17 cm in order to determine soil bulk density and coarse fraction.

Physical and chemical soil analysis

In the laboratory, all soil samples were dried at 40 °C until a constant weight was reached. Besides the detailed characterization of the soil organic matter composition, common soil properties were analyzed. The weight, volume and bulk density (BD) of the fine earth (< 2mm) were determined by dry sieving and water displacement of the coarse material (> 2mm). pH-value [-] was measured in the soil solution (1:2.5) with calcium chloride (0.01 M $CaCl_2$) (FAL, 1996). For plant growth essential soil macronutrients (Na^+, K^+, Mg^{2+}) were leached by the CO_2 extraction (1:2.5 soil solution) (FAL, 1996) and further determined by ionchromatography (Metrohm, 761 Compact IC). The concentrations of these nutrients were given per 100 g dry soil material (< 2mm) [mg 100 g^{-1} soil].

SOC fractionation

The responses of SOC stability to land cover changes can be investigated through physical fractionation methods, which allow to study the organization of OM within the soil matrix (Cambardella and Elliott, 1992; Six et al., 1998; Poeplau and Don, 2013; Hunziker et al., 2017). We therefore slightly modified the separation procedure presented by Zimmermann et al. (2007b) and used the disaggregation, the particle size separation and the density fractionation to separate the SOC into the given functional groups (von Lützow et al., 2007). The fractionation procedure was performed with the material of the fine earth (< 2mm) of the 69 samples. Initially, the samples were dispersed by an ultrasound treatment (22 J ml^{-1}) in 150 ml deionized water to retrieve only primary organo-mineral complexes (Christensen, 2001). Afterwards, the samples were wet sieved at 63 microns to separate the stable sand-sized aggregates and the un-protected particulate organic matter from the soil material < 63 microns. Subsequently, the particulate organic material (POM) was separated from the denser organic material in the mineral-associated sand and aggregate fraction (heavy fraction; HF) by density fractionation (1.8 g cm^{-3}, SPT from Sometu) on the soil material (> 63 microns). After separation, both fractions were washed with deionized water until the electrical conductivity of the rinse water reached < 50 µS (Wagai et al., 2008). The material < 63 microns represents the SOC pool of the silt and clay size fraction (S+C). Further, a sample of the suspension (< 63 microns) was taken after settling time, filtered with a 0.45 microns filter and analyzed for its dissolved organic carbon content (DOC). The value of the DOC concentration was used as an indication for the ability of the studied soils to leach dissolved organic carbon. Compared to Zimmermann et al. (2007b), the present study did not conducted the oxidation with sodium hypochlorite (NaOCl).

All samples of the bulk soil (< 2mm) and the soil fractions were ball-milled and then analyzed for organic carbon content by the dry combustion technique (Leco CN 628 Elemental Determinator). The dissolved organic carbon was measured by the combustion analytic oxidation method (TOC-5000A, Shimadzu).

The amount of soil organic carbon that is stored in a given soil profile is defined as the SOC stock and is expressed in tons per hectare. According to Ellert et al. (2008) and Rodeghiero et al. (2009), the SOC stock (SOCstock; t C ha^{-1}) is a function of the soil's carbon concentration (SOC_{conc}; [mg g^{-1}]), the bulk density (BD; g cm^{-3}) of the fine earth (< 2mm) and the investigated soil depth (l; cm). The conversion factor between the units is 100.

$$\mathrm{SOC_{stock}} = \mathrm{SOC_{conc}} \times \mathrm{BD} \times \mathrm{l} \times 100 \qquad \text{(Eq. 1)}$$

Characterization of the vegetation patterns

Within the 10 m by 10 m plot, we further recorded the vegetation based on the species composition and the hierarchic composition of the growing species according to Braun-Blanquet

(1964). Based on the vegetation mapping, 1-3 sub-plots were placed within the plot to conduct a woody biomass inventory for birch plants. The size of the sup-plots varied between 1 and 4 m^2, depending on the stem density. Within each sub-plot, the diameter (over bark) of all stems was measured at a length of 50 cm (Snorrason and Einarsson, 2006; Hunziker et al., 2014). In the case of branching below 50 cm stem length, the diameter was also recorded at branch length of 50 cm. The total biomass [t ha^{-1}] of the inventoried birch plants was estimated with the allometric equation for the above- and below-ground biomass of Icelandic mountain birch (*Betula pubescens* Ehrh. spp. czerepanovii) published by Hunziker et al. (2014). The chosen function is given for diameter ranges between 0.2 and 14.1 cm which corresponds to the stem diameters found in the present study (Tabelle 12).

Estimation of the duration of birch growth

The age of the birch stand was estimated by dendrochronology to provide an estimation of the least duration of the carbon transfer from the biomass into the soil. Assuming that the thickest stem in the inventoried sub-plot represents the oldest stem, we took a stem disk or collected a tree ring core of the thickest stem at each birch test site. Knowing the age of the oldest shrub, the minimum duration of shrub cover on the site can be deduced. This approach, however, ignores that younger stems of bushes can be thicker than older stems due to local site conditions which are more suitable for plant growth. This can be light, climate, competing conditions and availability of soil nutrients or water. The dataset of the wood samples does not cover all shrub sites due to the fact that samples were decayed to some extent or were broken during transportation.

In the laboratory at the Swiss Federal Research Institute for Forest, Snow & Landscape (WSL) in Birmensdorf, the samples were sledged with a microtome to receive 10 to 20 micrometer-tick sections which were colored with safranine as well as Basic Blue 140 and hardened with Canadian balm (Gärtner and Schweingruber, 2013). Subsequently, the microsections were digitalized by a microscope which was able to take pictures. The pictures of the subsections were stitched together with the software PTGui (New House Internet Services BV 2010). The program Win Dendro (Regent Instruments 2009) was applied to count the tree rings while the minimum and maximum radii were analyzed. The cross-validation of the data was performed with the software Tsap (Time Series Analysis Version 4.69d) (Gärtner and Schweingruber, 2013). The annual growth rate [mm yr^{-1}] was calculated based on the sum of the distances between all tree rings divided by the counts of tree rings. After the microscopic wood analysis, site specific growth patterns were observed at several wood samples. These ring anomalies in form of wedging rings and eccentric growth can be the result of insufficient solar radiation, nutrient supply and insect attacks or mechanical stress due to wind, snow or slope gradient (Schweingruber, 1996).

Surface air temperature recording

During the field campaign in 2014, the surface temperature was recorded by four data loggers (OM-EL-USB; OMEGA Engineering inc.) within the three sub-areas between the 13th and the 28th of July (Abbildung 21). The four loggers were installed close to the ground surface to a piece of wood inopen vegetation to represent the temperature characteristics of gramminoid tundra vegetation which can be potentially overgrown by shrub flora. In the sub-area “Valley”, the surface temperature was recorded in the valley bottom at 44 m a.s.l. and at the slope on the top of the valley at 250 m a.s.l.. The loggers on the “West” plateau and on the “East” plateau were placed at an elevation of 262 and 244 m asl, respectively. Another data logger was installed in the village of Igaliku on an elevation of about 10 m asl. Additionally, the available and preprocessed data of the synoptic weather station at Narsarsuaq which is located about 15 km north of the study area was used (Cappelen et al., 2015). The data of the temperature from Igaliku and Narsarsuaq were used for comparability reasons in regard to differences concerning the proximity to the sea water (logger “Igaliku”) and accuracy of the loggers compared to official temperature recording standards (“Narsarsuaq”).

The surface air temperature was recorded half-hourly. The records of the surface air temperature [°C] were used to calculate the descriptive statistics (minimum, mean, standard deviation and maximum values) of the “daily temperature” and the “daily magnitude”. In order to describe the daily temperature patterns for the period 13.07.-28.07.2016, the recorded temperature of the main standard synoptic hours (00, 06, 12 and 18 UTC) were used to calculate the statistics. For the same period, the statistics for the “daily magnitude” were calculated based on the ranges between the minimum and maximum values of all recordings on daily scale.

5.3 Results and Discussion

5.3.1 The quantitative and qualitative analysis of the soil organic carbon

In general, the SOC stocks (0-30 cm) of the birch sites varied between 54 and 148 t C ha^{-1}, while the birch site with the highest SOC stock was found on a straight-shaped, east-facing back slope in the “Valley” (Abbildung 23, Tabelle 11). The median SOC stock (95 t C ha^{-1}) of the sites at the western ridge was in the same range as the SOC stock (91 t C ha^{-1}) found in the valley (Tabelle 11), which was not expected because of the lower woody biomass stock (Tabelle 12). The SOC stock at the birch sites on the eastern ridge revealed, however, considerably lower values (median: 73 t C ha^{-1}). Therefore, the ranking of the SOC stocks of the birch sites was “Valley” ≥ “West” > “East”. Compared to the shrub-free sites which represent the majority of the vegetation cover of the landscape unit (west, valley east), the birch sites contained higher SOC stocks (0-10 cm, 0-30 cm) in the “Valley” and on the eastern plateau. Equal or higher SOC stocks (0-10 cm) were however measured for the shrub-free sites at the western ridge

(Abbildung 23, Tabelle 11). The SOC stocks of the shrub-free sites can be ranked as “Valley” > “West” ≥ “East”. According to Jobbágy & Jackson (2000), the estimated SOC stock (0-30 cm) for the Boreal and Tundra biome is approximately 37 t C ha^{-1} (N = 648) and 50 t C ha^{-1} (N = 51), respectively. In comparison with the present study, the SOC stocks of the birch sites which represent the transition zone between Boreal and Tundra biome, as well the un-woody sites which represent the Tundra biome, showed considerably higher values than given in Jobbágy & Jackson (2000). Köchy et al. (2015) reported about 100 t SOC ha^{-1} for soils along the Greenlandic coastlines. The circumpolar soil atlas reports SOC stocks of 10-50 t C ha^{-1} for Leptosols, 100-200 t C ha^{-1} for humus-rich Cambisols, 150-200 t C ha^{-1} for Podzols and 250-400 t C ha^{-1} for deep and humus-rich arctic Cambisols which also can be found in southwest Greenland (Jones et al., 2010). In contrast to the circumpolar soil atlas, which indicates SOC stocks measured to the bedrock, the presented study focused on the first 30 cm (Abbildung 23), which is also recommended by the Aalde et al. (2006a). This could explain the different results. The comparison with available data from the literature shows that comparisons of SOC stocks are difficult and disputable due to the fact that the referred data represents the SOC stock on landscape units of different scales (Jobbágy and Jackson, 2000) or is given for a different soil depth (Jones et al., 2010). Further on, the results are often based on interpolations techniques which implicate a certain bias of the chosen models (Hiederer et al., 2011). On the other site, the bias of regional or global models needs to be reduced to estimate carbon pools more reliable (Canadell et al., 2010). This calls for a denser network of field data which should be repeated regularly (Canadell et al., 2010).

Field studies on SOC stocks for Greenlandic soils are sparse. Thus, the Greenlandic greenhouse gas inventory even utilizes the IPCC default values (Nielsen et al., 2011), which vary between 71 and 115 t C ha^{-1} given for different soil types in the cool, temperate and moist climate (Aalde et al., 2006a). Recently, several studies, however, examined the SOC dynamics in Greenland. Henkner et al. (2016) investigated the SOC patterns of permafrost-affected soils in the glacial forefield of the Ørkendalen glacier (near Kangerlussuaq, West Greenland). Amongst other landscape types, they reported SOC stocks (0-30 cm) for the valley bottom and the crest of hilltops or moraine fields of 142 t C ha^{-1} and 60 t C ha^{-1}, respectively (Henkner et al., 2016). The values of the present study are in the range of those numbers. Another study, also conducted in the Kangerlussuaq area, found lower mineral SOC stocks (0-60 cm) under shrub vegetation (225 t C ha^{-1}) than under graminoid vegetation (290 t C ha^{-1}) (Petrenko et al., 2016). A former study, addressing the suitability of Greenlandic soil for hay production, analyzed physical and chemical soil properties on agriculturally used fields and unused reference sites just nearby the presented study area. For both, the reference sites and the agricultural used areas, the median SOC stocks (0-30 cm) was equal on a level of 177 t C ha^{-1} (Caviezel et al., in prep.). The higher C stocks can thereby be explained by land use history of the analyzed areas. The Vikings already cultivated parts of the fields which are still in use today (Arneborg, 2005; Massa et al., 2012). To enhance the fertility and probably also to reduce the duration of snow cover on the parcels, they used manure (Arneborg, 2005; Bichet et al., 2013; Vésteinsson et al., 2014) and seaweed (Gröntved, 1956) beside let grazing the domestic animals on the fields (Ross et al., 2016).

The literature review revealed that the present study analyzed the SOC quality of Greenlandic soils for the first time. However, the analysis of the SOC characteristics needs to be implemented when the SOC is studied in relation to the change of C input into the soil due the land cover change (Jandl et al., 2014). The SOC fractionation procedure disclosed that the most relevant SOC pools were the POM fraction and the S+C fraction (Abbildung 24, Tabelle 11). Over all three sub-areas, the SOC fractionation by depth and physical separation techniques revealed in general that the relative amount of SOC in the POM fraction was higher in 0-10 cm than 0-30 cm while the opposite trend was visible for the SOC in the S+C fraction (Abbildung 24). The birch stands in the valley showed even higher POM-SOC stocks (median: 51 %). The relative amount of SOC in the HF fraction was equal distributed in these two depth intervals. The SOC in the HF fraction showed the lowest variation independently of vegetation type or location (Tabelle 11). Less than 20 % of the SOC stock was sequestered in the HF fraction (sand + aggregates). The amount of carbon which can be transported to the aquatic system (DOC) was on a very low level in all tested categories (< 2 % of the total SOC stock) (Abbildung 24, Tabelle 11).

Within the sub-areas, the SOC (0-30 cm) at the shrub sites in the “Valley” was mostly stored in the POM fraction. In the “Valley”, the test sites T2 S4 and T2 $S3Ref_{30}$ were located next to each on an alluvial fan, while the shrub-free site (T2 $S3Ref_{30}$) was completely covered by grass vegetation. According to the measurements, both sites contained a similar SOC stock (0-30 cm) (Abbildung 23). Concerning the stocks in the SOC fractions, the shrub sites however contained 46 and 34 t C ha^{-1} in the POM and S+C fractions, respectively, while the graminoid site stored 26 and 46 t C ha^{-1} in the POM and S+C fractions (Tabelle 11). In the sub-areas “West” and “East, similar patterns as in the “Valley” were observed however less distinct (Abbildung 24, Tabelle 11).We therefore assume that the ingrowth of shrub vegetation can lead to change of the SOC quality from more stable SOC in the S+C fraction to more labile SOC in the POM fraction without a considerable increase of the SOC stock of the mineral soil.

5.3.2 The carbon sequestration potential and stability in the soil

Differences between the SOC stocks of the bulk mineral soil between shrub vegetation and shrub-free vegetation can be used to estimate if the soil acts as C-source or C-sink during the shrubification of tundra grassland. According to Petrenko et al. (2016), the SOC stocks (0-60 cm) under shrub vegetation was lower (65 t C ha^{-1}) compared to the SOC stock under graminoid vegetation (290 t C ha^{-1}). In the present study, the SOC stock of the shrub vegetation was higher than that of shrub-free vegetation (Tabelle 11) which is most pronounced at the sub-area “East” (“East” > “Valley” >> “West”). In consequence, the conversion of tundra vegetation into shrub vegetation would lead to a minor increase of the total SOC stock which most noticeable on the eastern plateau. However, the results showed that on these sites an ingrowth of shrub vegetation would mainly increase the SOC in the POM fraction (Tabelle 11).

5.3.3 Plant biomass and productivity as a controlling factor for SOC patterns

The differences of the SOC stocks in the bulk soil and in the SOC fractions between the different vegetation types as well as the different sub-areas might be explained by the availability of C material as soil organic carbon is influenced by the growing vegetation and its litter productivity (Guo and Gifford, 2002; Vesterdal et al., 2013). Shrubification within the boreal-tundra ecotone is expected to increase the soil carbon stock due to the higher net primary production of shrub communities (Tømmervik et al., 2009). Biomass productivity, indicated by the silvicultural stand characteristics (Tabelle 12), showed distinct differences between the analyzed birch sites in the three sub-areas. The tree inventory revealed that the birch shrubs at “West” contained the thinnest (max. 1.2 cm) diameters [D_{50}] and the shortest growth heights [H_d] (max. 1.5 m) compared to the two other sub-areas (Tabelle 12), where the thickest (max. 9.4 cm) and highest (max. 3.6 m) bushes were found in the “Valley” (Tabelle 12). In spite of the clearly highest diameter values at “Valley”, the dendrochronological analysis found similar ages for all three sub-areas, but varying growth rates, which can be seen as an indicator for stand productivity (Tabelle 12). The revealed annual growth rates between 0.21 and 0.59 mm are comparable those from Fennoscandinavia (Treter, 1984; Karlsson et al., 2004).

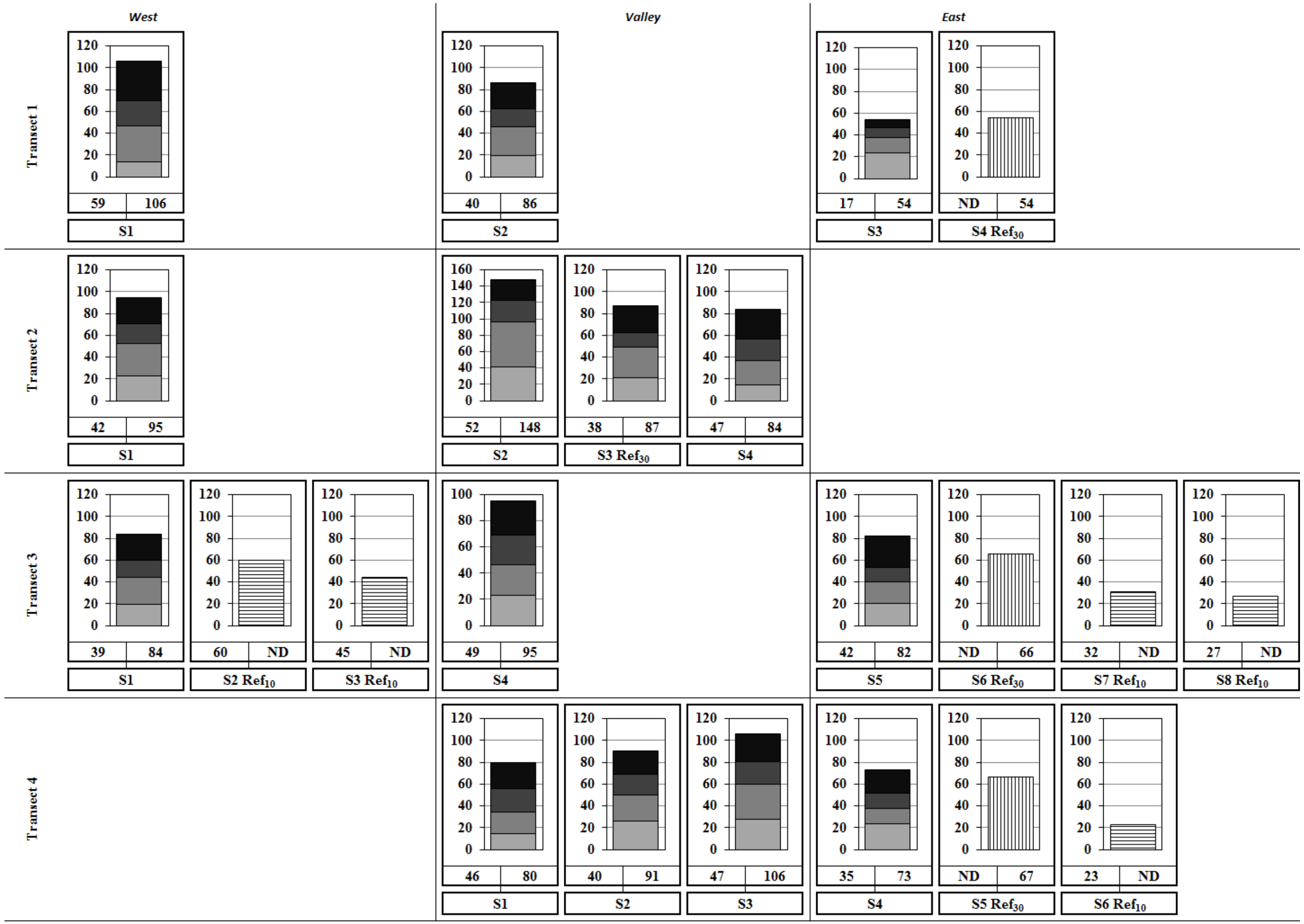


Abbildung 23: Bulk SOC stocks [t C ha-1] of the shrub sites and the shub-free sites (striped bars) (left: 0-10 cm, right: 0-30 cm) in numbers. Further, the stocks of the shrub sites are given for the sampled soil intervals (0-5 cm: black, 5-10 cm: dark grey, 10-20 cm: grey, 20-30 cm: light grey). The striping at the shrub-free sites indicates the sampling depth (horizontal: 0-10 cm, vertical: 0-30 cm). The SOC stocks are grouped according to the four transects and the three sub-areas.

Abbildung 24: The proportion [-] of the SOC concentrations (striped: POM, checked: HF, dotted: S+C, black: DOC) within the tested soil intervals (left: 0-10 cm, right: 0-30 cm). The bulk SOC stocks [t C ha-1] are given for 0-10 cm and 0-30 cm. The data is grouped according to the four transects and the three sub-areas.

Tabelle 11: Median SOC stock [t C ha^{-1}] of the bulk soil (< 2 mm) and the analyzed fractions (POM, HF, S+C and DOC) for 0-10 and 0-30 cm soil depth. The parentheses show the minimum and maximum values. In the case of N < 7, the raw data is showed.

Location/ Type	SOC stock [t C ha^{-1}]									
	Bulk		POM		HF		S+C		DOC	
	0-10 cm	0-30 cm	0-10 cm	0-30 cm	0-10 cm	0-30 cm	0-10 cm	0-30 cm	0-10 cm	0-30 cm
West										
Birch (N_{10} = 3; N_{30} = 3)	42 (39; 59)	95 (84; 106)	23 (14; 38)	39 (24; 55)	8 (8; 9)	18 (18; 18)	13 (11; 16)	37 (31;40)	0.6 (0.5; 0.7)	1.3 (1.2; 1.6)
Reference (N_{10} = 2; ND)	45;60	-	19; 25	-	3; 7	-	18; 31	-	0.8; 1.0	-
Valley										
Birch (N_{10} = 7; N_{30} = 7)	47 (40; 52)	91 (80; 148)	25 (21; 36)	46 (31; 91)	5 (2; 9)	14 (7; 21)	13 (9; 17)	34 (24; 41)	0.8 (0.4; 1.1)	1.8 (0.9; 2.3)
Reference (N_{10} = 1; N_{30} = 1)	38	85	14	26	4	13	19	46	0.7	1.9
East										
Birch (N_{10} = 3; N_{30} = 3)	35 (17; 42)	73 (54; 82)	18 (6; 23)	28 (20; 37)	5 (4; 5)	13 (11; 14)	12 (7; 14)	30 (23; 31)	0.5 (0.3; 0.6)	1.1 (0.8; 1.2)
Reference (N_{10} = 3; N_{30} = 3)	27 (23; 32)	66 (54; 67)	7 (5; 16)	23 (16; 25)	5 (4; 6)	10 (9; 11)	13 (10; 14)	30 (27; 33)	0.6 (0.5; 1.0)	0.9 (0.3; 1.7)

According to these findings, the estimated basal area [m^2 ha^{-1}] as well as the total (above- and belowground) biomass [t ha^{-1}] were considerably higher at most of the “Valley” sites of compared to those of the other two sub-areas. The highest living biomass was calculated for the sub-area “Valley” with the highest estimated biomass pool (182.3 t ha^{-1}) at site “T2 S2”. The median biomass pool was considerably lower at “West” (8.6 t ha^{-1}) and “East” (21.8 t ha^{-1}) than at “Valley” (59.2 t ha^{-1}). Elkington & Jones (1974) measured 54.9 t ha^{-1} for living above- and below-ground biomass for mountain birch woodlands on sheltered slopes about 30 km southwest of Igaliku by applying a partially destructive sampling technique. This reference value is most comparable with the median biomass estimation for the “Valley” compared to the biomass values found in the other sub-areas (Tabelle 12). However, the present study found much higher values in the “Valley" for distinct sites. Futher on, the biomass estimations for the birch stands on the plateaus “West” and “East” showed much lower values with one exception (about 110 t ha^{-1}) at the sub-area “East”.

The study showed that the silvicultural properties like dominant height, basal area and biomass pool of the birch stands differed between the three sub-areas. This is a plausible reason for the differences in the SOC stock between the birch stands of the different sub-areas. In the following, we want to find explanations for the abundance and growth patterns of mountain birch shrubs on landscape level. The identified controlling factors can be further used to improve climate-vegetation models (Canadell et al., 2010; Normand et al., 2013).

5.3.4 Possible controlling factors for shrub growth and SOC patterns

Temperature

Knowing that the given time interval (13th to 28th July 2014) of one single year is to short and statistically not representative to detect any seasonal or annual changes, the data can be used to identify differences of air temperature within a catchment at the same time during the warmest month of the year. Within this time span, differences in the summer air temperature were measurable between the three sub-areas (Tabelle 13). On a daily scale, the highest mean and maximum temperatures were recorded on the top of the Valley. Further, the results revealed the highest mean and maximum daily magnitude (range between minimum and maximum temperature within a day) at the top of the valley. The temperature reached three times over 35 °C at the top of the valley (Tabelle 13). The reason for the highest maximum values found at “Valley_top” might be caused by local heating effects at the valley slopes and hence ascending up winds through the valley towards the recording station

Tabelle 12: Silvicultural characteristics of the birch sites separated by the three sub-areas “West”, “Valley” and “East”. The data is given by the defined woody vegetation layer according to Braun-Blanquet (1964). Besides the percentage [%] of the vegetation cover, the mean diameter at 50 cm stem or branch lengths [cm], the mean length of the stems or branches [m], the dominant growth heights [m], the mean basal area [m^2 ha^{-1}], the estimated above- and below-ground biomass [t ha^{-1}], the age and the annual growth rate [mm yr^{-1}] are displayed for each test site. The coverage [%] was used to adjust the estimations for the basal area and total biomass.

Sub-Area	Test Site	Vegetation Layer	Birch coverage [%]	D_{50} [cm]	H_d [m]	BA [m^2 ha^{-1}]	Biomass [t ha^{-1}]	Age [yrs]	Growth Rate [mm yr^{-1}]
West	T1 S1	1	35	0.7	1.4	3.7	4.8	40	0.34
		2	15	0.9	0.8	2.5	3.8	Nd	Nd
	T2 S1	1	30	1.0	0.8	3.9	5.9	26	0.41
	T3 S1	1	40	1.2	1.5	7.9	14.8	40	0.21
Valley	T1 S2	1	30	2.5	2.5	24.5	54.4	64	0.58
		2	20	1.9	1.3	2.9	4.8	Nd	Nd
	T2 S2	1	45	9.4	3.6	41.1	120.6	57	0.40
		2	45	6.8	1.8	22.5	61.7	Nd	Nd
	T2 S4	1	90	4.6	3.0	51.3	123.4	Nd	Nd
	T3 S4	1	80	3.4	2.6	41.2	89.5	Nd	Nd
	T4 S1	1	45	0.7	1.1	3.4	4.5	24	0.48
	T4 S2	1	65	2.7	1.8	17.3	37.9	50	0.59
		2	15	0.7	1.0	0.3	0.4	Nd	Nd
	T4 S3	1	80	1.4	1.3	9.8	32.9	36	0.45
	T4 S3	2	10	1.5	0.5	2.0	3.7	59	0.33
East	T1 S3	1	65	1.7	1.8	12.5	21.8	44	Nd
	T3 S5	1	70	3.3	2.0	46.6	103.0	57	0.51
		2	15	1.7	1.5	3.3	6.3	60	0.47
	T4 S4	1	20	2.0	1.8	3.0	5.4	39	0.26
		2	60	0.8	1.5	3.3	4.5	Nd	Nd

"Valley_top". During night time which is given by daily minimum temperatures, the differences were smaller than during day time (12:00 UTC) when the highest values were measured.

Based on this, it seems that the topographic setting of the study area influences the temperature characteristics. Another finding was that the mean and maximum temperatures as well as magnitudes on daily scale were lowest in the sub-area "East". Several studies found that summer temperatures especially in July strongly affects the stem and plant growth of mountain birch in northern Iceland (Levanič and Eggertsson, 2008), Sweden (Kullman, 1993; Karlsson et al., 2004) and Greenland (Kuivinen and Lawson, 1982; Jørgensen et al., 2015). Sigurdsson (2009) measured the diameter increment of mountain birch in southern Iceland and found that the increase of the diameter occurs between the 1st June and 1st September with the highest increment rates between mid of June to end of July.

Böcher (1979) who found higher summer air temperatures at birch stands compared to sites with shrub-free vegetation. The present study confirmed this finding by higher mean daily temperatures (> 1 °C) at protected sites in the "Valley" compared to the other sub-areas. Hence, temperature seems to be a relevant controlling factor on local scale for the abundance and the growth patterns of birch vegetation and thus for the allocation of carbon in the biomass and in the soil.

Tabelle 13: Surface air temperature recordings between the 13th and the 28th July 2014.

Sub-area, elevation	Daily temperature [°C]				Daily magnitudes [°C]			
	Min	Mean	StDev	Max	Min	Mean	StDev	Max
West (262 m asl)	6.5	14.2	7.3	24.0	7.5	23.1	7.0	34.0
Valley_bottom (44 m asl)	6.2	15.6	8.5	26.1	12.0	24.3	6.2	34.5
Valley_top (250 m asl)	5.8	15.9	9.2	27.4	10.5	27.0	6.6	35.5
East (244 m asl)	6.6	14.5	7.3	23.0	6.0	22.3	6.0	29.5
Igaliku (10 m asl)	8.8	14.6	5.6	21.8	5.5	17.8	5.1	26.5
Narsarsuaq (27 m asl)	8.3	11.7	2.7	15.6	3.0	8.0	2.5	11.5

Substrate and bedrock material

The sub-areas were defined according to geologic settings of the study area (Sørensen et al., 2006). The bedrock material consisted of sandstones belonging to the Eriksfjord Formation in the "West", nepheline syenite as intrusion material from the Igaliko Complex in the "East" and Julianehab granite in the "Valley". In the "Valley", the parent material however consisted of fluvio-glacial transported material of nepheline syenites and sandstones. In the "West" and "East", the parent material dominantly consisted of weathered and eroded bedrock material. Due to the different bedrock and parent material types at the different sub-areas, the soil analysis found differences in the coarse fraction ratio [%] (Tabelle 14). The soil material at "East" contained the highest amount of coarse material (> 2 mm) (between 13 and 45 % of mass weight). This can be attributed to the coarse grained texture of the plutonic rock material and the weak stability concerning physical weathering processes which was observed in the field. At “West” and in the “Valley”, the coarse fraction ratio was lower than 10 %. Only, “T2 S4” and “T4 S1” in the “Valley” showed slightly higher values than 10 % which is attributed to geomorphic transported material like rockfall deposits at “T4 S1” and deposits at the beginning of a fluvial fan at “T2 S4” (Tabelle 14). In general, the soils were acid with slightly higher soil pH values at “East”. Experiments performed by Ingestad (1979) showed that the maximum growth rate of birch seedlings is between pH values 4.0 to 6.8. Hence, birch plants can establish on these parent materials with regard to the pH value. The measured pH values ($CaCl_2$) also indicate a low base saturation. For all tested soil macronutrient, the concentrations [mg 100 g^{-1} dried soil] were highest at the test sites of the “Valley” (Tabelle 14). Also, the concentrations of potassium which is mostly the limiting soil nutrient concerning plant growth (Wang et al., 2013)were considerably higher at “Valley” than at “West” and “East”. This can be another explanation for the higher biomass production (Tabelle 12) and hence the higher SOC stocks (Tabelle 11) in the “Valley” than on the plateaus. At “East” and “West”, the reference sites showed higher soil nutrient concentrations than the tested birch sites which were considerably higher in some cases. Hence, there might be other factors which can hamper the establishment and growth of shrub flora.

Tabelle 14: Physical (coarse fraction ratio of mass weight, density fine earth) and chemical (pH value, cation (Na^+, K^+, Mg^{2+}) concentrations) properties of all tested birch (B) and shrub-free (R) sites. Sampling depths of only 0-10 cm instead of 0-30 cm are indicated by a star (*).

Sub-Area	Test Site	Type	Coarse Fraction Ratio [%]	Density Fine Earth [g cm^{-3}]	pH ($CaCl_2$) [-]	Na^+ [mg 100 g^{-1} soil]	K^+ [mg 100 g^{-1} soil]	Mg^{2+} [mg 100 g^{-1} soil]
West	T1 S1	B	8.8	0.74	5.3	1.6	3.0	1.7
	T2 S1	B	6.1	0.66	4.7	1.6	2.1	1.2
	T3 S1	B	3.4	0.78	4.7	1.7	1.6	0.9
	T3 S2*	R	2.1	0.51	5.0	5.5	8.2	2.0
	T3 S3*	R	5.7	0.48	4.8	5.7	6.2	2.5
Valley	T1 S2	B	8.8	0.76	5.7	2.5	2.1	2.1
	T2 S2	B	2.2	0.52	Nd	6.1	21.7	5.7
	T2 S3	R	8.0	1.01	Nd	8.1	12.0	5.7
	T2 S4	B	15.6	0.88	Nd	10.3	17.2	7.8
	T3 S4	B	3.2	0.67	Nd	10.4	15.4	5.4
	T4 S1	B	12.3	0.71	Nd	3.8	4.7	2.8
	T4 S2	B	9.8	0.71	Nd	7.3	5.6	4.2
	T4 S3	B	4.9	0.75	Nd	6.4	11.4	6.0
East	T1 S3	B	13.7	1.00	5.4	1.7	3.1	1.3
	T1 S4	R	18.1	1.12	5.6	1.5	3.8	2.0
	T3 S5	B	23.5	0.84	5.4	1.8	2.8	1.9
	T3 S6	R	25.6	0.90	5.6	2.8	4.5	2.3
	T3 S7*	R	20.6	0.86	Nd	4.6	10.9	6.1
	T3 S8*	R	18.3	0.84	Nd	4.5	4.8	3.0
	T4 S4	B	34.4	0.69	5.0	1.5	2.5	2.2
	T4 S5	R	29.9	0.92	5.2	1.9	3.0	2.5
	T4 S6*	R	45.0	0.90	5.4	2.3	6.1	4.0

An assemblage of spatial covariances derived from relief

Landscapes are formed by different endogenic and exogenic forces which affect on the terrestrial surfaces consisting of different upcoming bedrock material. Hence, landscapes are unique by containing a patchwork of landscape elements which can be characterized by a number of topographic features. Hence, these features are also used to describe the location of vegetation or soil types (FAO, 2006b). The importance of the relief and its spatial covariances concerning the SOC patterns within the landscape was showed in several case studies (Burke et al., 1999; Yoo et al., 2006; Berhe et al., 2008; Garcia-Pausas et al., 2008; Hancock et al., 2010).

In most cases, the land surface of sampled sites and the nearby surrounding was gently sloping (slope angle: 2-5 %) at “West” and “East” and sloping (slope angle: 5-10 %) at “Valley” (Tabelle 15) which favors the water infiltration compared to surface runoff. Independently of the sub-area, the sites were south-east, south or south-west orientated (Valley: 1 north-west). In the “Valley” and at “East”, the sites were mostly located at middle slopes or in accumulations zones (LS, TS) and the slope lengths above the sites were mostly between 100 and 200 m. In contrast to this, the slope lengths above the sites at “West” were maximum 30 m long due to the nested location of the vegetated patches within the glacial shaped landscape (Abbildung 20) which was characterized by sandstone as bedrock (Sørensen et al., 2006).

The slope positions of the birch sites in the “Valley” might be also responsible the SOC stocks which were higher in deeper sampling intervals (10-20 cm, 20-30 cm), in some cases (Abbildung 23). In some cases, birch sites rather showed higher SOC stocks in the sampling layers 10-20 and 20-30 cm. Besides the doubling of the SOC pool by the investigated soil thickness, the increase of the SOC stock is more attributed to the increase of the bulk density due to the general decrease of the SOC concentration (data not shown). This can be argued by the position of the sites on the slopes (Tabelle 15). Except of test site T3S8 (UP), all sites were positioned at middle slope (MS), lower slope (LS) or toe slope (TS) which are position where sedimentation especially under vegetation (MS) can occur. The relatively long up-slope distances and the mostly straight and concave slope forms indicate the sedimentation potential at the test sites (Tabelle 15). Hence a possible reason for the higher SOC stock in deeper soil layers can be the lateral transportation due to soil erosion and the sedimentation of fine earth material in historic times which caused denser soil horizons. Another argument for the vertical SOC patterns is the vertical transportation of SOC material due to podsolization which occurs in densely vegetated areas (Feilberg, 1984). The laterial transportation of material into the test sites might be reasonable for the highest nutrient concentrations in the “Valley” compared to the two other sub-areas (Tabelle 14, Tabelle 15).

The relief and the topographic setting can influence the features which were discussed above. Depression zones like valleys and U-shaped channels (at “West”) can act as traps for mineral material, organic substances like litter, water or snow during wind events (Abbildung 20).

Chapin et al. (1995) showed that the snow cover can increase the soil surface temperature by 3 to 10 °C. This stimulated the soil biota and hence the mineralization rate of nitrogen which enhanced the plant nutrient availability (Chapin et al., 1995). Higher soil temperatures can also be caused by higher air temperatures due to climate warming or microclimatic prime sites in depression zones (Tabelle 13). The present study did not measure precipitation which was showed to be relevant for tree ring growth in form of precipitation during spring (Levanič and Eggertsson, 2008). However, relief controls further the availability and the amount of water. At the sub-area “West”, depression zones accumulate and retain surface water hence the formations of ponds, swamps and the associated vegetation community containing *Vaccinium uliginosum Anthoxanthum nipponicum* are the consequences. The accumulation of water was assisted by the hampered infiltration capacity into the bedrock consisting of sandstone (Sørensen et al., 2006). Shrub vegetation was found on sites with drier soil conditions. Due to the strongly hampered decomposition of organic soil material, the SOC stock of the shrub-free sites were slightly higher than at the sites with shrub vegetation at “West”. In contrast to this, the topographic setting of the “Valley” favors the accumulation of runoff water during rainfall events and the snowmelting time. Due to the soil substrate, the slope angle and slope length, water can percolate laterally through the soil matrix and water stagnation can be expulse (Tabelle 15). At the sub-area “East”, the high amount of coarse fraction consisting of plutonic material causes a high infiltration rate which can be limited for shallow rooting species like mountain birch.

Elevation is another topographic measure which controls the vegetation patterns within the landscape (Körner, 2012). Böcher (1979) stated the “birch forest” and the “krummholz” treelines at elevations of 150 and 250 m asl, respectively. This can be confirmed by the present study which found considerable smaller birch bushes on the higher elevated sub-areas “West” and “East” than in the “Valley” (Tabelle 12). The harsher climate at higher elevation limits the primary production and hence the C input to the soil (Djukic et al., 2010). On the other site, the decomposition of the SOC is hampered at higher elevation due to the lower microbial activity which is attended by soil temperature at higher elevation (Simmons et al., 1996).

Tabelle 15: Topographical site characteristics according to FAO (2006b).

Sub-Area	Test Site	Type	Elevation	Slope characteristics				
				Angle	Orientation	Position	Form	Length
			[m asl]	[°]	[-]	[-]	[-]	[m]
West	T1 S1	B	215	13	SE	LS	SS	20
	T2 S1	B	270	15	SW	MS	CS	10
	T3 S1	B	265	20	SE	MS	SC	20
	T3 S2	R	246	8	S	MS	SC	30
	T3 S3	R	250	14	W	MS	SV	30
Valley	T1 S2	B	90	4	SW	TS	SS	100
	T2 S2	B	138	24	SE	MS	SS	150
	T2 S3	R	84	10	SE	LS	SS	160
	T2 S4	B	72	12	SE	LS	VV	150
	T3 S4	B	54	13	SW	MS	VS	120
	T4 S1	B	131	19	SE	MS	SS	70
	T4 S2	B	143	27	NW	MS	VC	150
	T4 S3	B	204	24	S	MS	VV	30
East	T1 S3	B	124	11	W	MS	SS	200
	T1 S4	R	130	21	W	MS	SS	200
	T3 S5	B	191	11	SW	LS	SS	100
	T3 S6	R	212	20	SE	MS	SS	100
	T3 S7	R	219	15	E	MS	SS	120
	T3 S8	R	221	13	SW	UP	SS	100
	T4 S4	B	240	12	SE	MS	SS	30
	T4 S5	R	245	8	SE	MS	SS	90
	T4 S6	R	241	15	E	MS	VV	100

5.4 Conclusion

The present study used the catena approach to investigate the SOC properties of mountain birch stands and sites which contained graminoid tundra vegetation. Based on this, the changes of the SOC quantity and quality were estimated with regard to the expansion of the shrub vegetation within the tundra-boreal ecotone induced by the recent and future climate warming. The results revealed that the SOC stocks (0-30 cm) varied between 54 and 148 t C ha^{-1}. The highest SOC stocks (0-30cm) 80 und 148 t C ha^{-1} were found on the sites situated on the valley bottom in the study site. The sites located above 200 m asl contained similar (84-106 t C ha^{-1}; West) or lower (54-82 t C ha^{-1}; East) bulk SOC stocks. Differences between the bulk SOC stocks of shrub and shrub-free vegetation were marginal. Besides elevation as a controlling factor for SOC stock, the amount of biomass corresponded positively with the SOC stocks. Soils under shrub vegetation stored considerably more carbon in the less stable POM fraction than soils under shrub-free vegetation. There, the SOC stock in the silt and clay fraction was higher under shrub-free vegetation. Thus, the study concludes that shrubification can increase the vulnerability of the SOC concerning its stability. However, shrubification in the boreal-tundra ecotone is not only related to the change of climate on regional scale. According to the present study, it seems that on local scale shrub vegetation responses to other key factors. The case study on catchment-scale was not able to find the dominant controlling factor for occurrence of shrub vegetation and thus increase of biomass production. However, the study was able to show a palette of features which influence the vegetation patterns within the boreal-tundra ecotone. The applied layer-based model approach revealed that lithologic and topographic characteristics of the landscape elements seem to be the major controlling factors on catchment-scale. The relief affects directly the properties of the shrub vegetation by elevation and aspect. Test sites with shrub vegetation were dominantly W, S and E exposed and the dominant shrub height was lower at sites on higher elevation. Indirectly, the spatial covariances "bedrock material" and "relief" are responsible for local landscape forms and hence differences in microclimatic and pedologic (e.g. coarse fraction, nutrient status) characteristics which influences itself the occurrence and growth patterns of shrub vegetation as well the storage of soil organic carbon.

Further, the field study reveals that the prediction of the expansion of shrub vegetation and its assumed carbon storage potential in the vegetation and the soil is based on regional-scaled climate-vegetation models is biased due to the overgeneralization to only climatic variables and the non-consideration of additional variables referring to terrestrial properties which influence the abundance and growth patterns of shrub vegetation and hence the change of the SOC properties on larger scale.

KAPITEL 6

Diskussion

Die Böden im Hekluskógar sind noch immer stark degradiert. Die Flächen sind vor der Beweidung geschützt und die Rekultivierung beginnt meist mit der künstlichen Düngung und der Ansaat von Grasarten, Strandroggen oder Alaska Lupinen. Am linken Bildrand fliesst ein Seitenarm des Fluss Ytri-Rangá um den Birkenbuschwald „Hraunteigur", welcher noch in den linken Bildteil ragt. Im Hintergrund ragt der Vulkan Hekla in eine Höhe von 1491 m ü. M. empor. Aufgenommen von M. Hunziker am 10. Juli 2009.

6.1 Kritische Beurteilung der Qualität der Datensätze unter Berücksichtigung des Evaluationsbogens von Jandl et al. (2014)

6.1.1 Beprobungsstrategie

Auswahl der Probeflächen

Der Chronosequenz-Ansatz bietet die Möglichkeit, Veränderungen des Bodenkohlenstoffes in Folge einer Landbedeckungsänderung zu dokumentieren. Die Annahme des Chronosequenz-Ansatzes ist, dass sich die Beprobungsstandorte nur in der Zeitdauer und der zu untersuchenden Variable(n) unterscheiden. Damit können Veränderungen in Ökosystemen, die natürlicherweise über Jahrzehnte ablaufen, räumlich nebeneinander studiert werden (Walker et al., 2010). Damit können Veränderungen während des Systemwechsels (SOC Verhalten bei Verbuschung) im Voraus prognostiziert werden. Die Auslegung der Beprobungsflächen auf Basis einer Chronosequenz benötigt eine detaillierte Vorstudie, welche die oben erwähnte Grundannahme bestätigt.

Die Grundidee der vorliegenden Arbeit war bei allen drei Fallstudien, die Bodenkohlenstoffveränderungen mit Hilfe des Chronosequenz-Ansatzes aufzeigen zu können. Dieses Vorhaben ist jedoch nur bedingt umsetzbar gewesen. In der Fallstudie „Aufforstung“ in Island hat sich bei der Analyse der Resultate gezeigt, dass die Böden der einzelnen Alterskategorien hinsichtlich ihrer Ausgangskohlenstoffkonzentration zu unterschiedlich sind und die Bedingung für den Chronosequenz-Ansatz eigentlich nicht gewährleistet ist. Die Böden der stark degradierten und unbewachsenen Flächen (Barren Land), die den Ausgangszustand darstellen (Halldórsson et al., 2009), weisen deutlich höhere Kohlenstoffkonzentrationen in den tieferen Beprobungsschichten auf als jene der aufgeforsteten Birkenstandorte mit einem Alter von 15 und 20 Jahren (Kapitel 3). Daher können die Flächen von „Barren Land“ nur schwer als Ausgangszustand für die Veränderung des Kohlenstoffvorrates durch die Aufforstung herangezogen werden. Bei der Fallstudie „Klimaerwärmung“ (Kapitel 5) sind mittels Luftbildvergleich während der Vorstudie keine sichtbaren Veränderungen in der quantitativen Ausdehnung der Buschvegetation im Untersuchungsgebiet erkennbar gewesen. Die visuelle Aufnahme der Vegetationseigenschaften während der Begehung im Jahr 2013 hat die Wahl des Catena-Prinzips als Beprobungsstrategie bestätigt. Laut Leser (2010) dient das Catena-Prinzip dem Erkennen von geoökologischen Raummustern, die sich in der Landschaft deutlich voneinander unterscheiden lassen. Die Probeflächen, die in der Fallstudie entlang von vier Transekten platziert worden sind, repräsentieren mit Busch- oder Tundravegetation bewachsene Standorte innerhalb der ausgeschiedenen Landschaftselemente.

Die Anwendung der beiden zuvor beschriebenen Beprobungsansätze bietet die Wahl des „stratified random sampling“ (Tabelle 16). Dabei repräsentierten die unterschiedlichen

Vegetationstypen resp. Alterklassen die Schichten (engl. strata) des „stratified random sampling". Die Auswahl der Beprobungsstandorte innerhalb der einzelnen Schichten erfolgte anschliessend per Zufall.

Beprobungstiefe

An den meisten Standorten sind die ersten 30 cm beprobt worden, was die gängige Beprobungstiefe bei Kohlenstoffinventaren ist und mindestens dem „Level 2" des Evaluationsbogens von Jandl et al. (2014) entspricht (Post and Kwon, 2000). Im Projekt „Klimaerwärmung" sind bei fünf Standorten mit Tundravegetation jedoch nur die obersten 10 cm des Bodens beprobt worden, was die Ausnahme darstellt. Bei der Erstellung eines Kohlenstoffinventars oder beim Monitoring von Böden spielen die drei Ressourcen „Einsatzkräfte", „Zeit" und „Finanzen" eine entscheidende Rolle für den Umfang des Datensatzes. Daher liegt der Fokus der Studien in vielen Fällen auf den obersten 30 cm des Bodens. Diese Beprobungstiefe bietet sich jedoch bei der Untersuchung des Bodenkohlenstoffes infolge des Landbedeckungswandels aufgrund des Eintrages von organischem Material über den darüber liegenden Auflagehorizont und die sich darin befindende Wurzelbiomasse als C-Quellen an. Aus diesen Gründen ist in den drei Fallstudien die Beprobungstiefe von 30 cm angewendet worden.

Bodenkohlenstoff wird aber auch in tieferen Bodenschichten als in den obersten 30 cm gemessen (z.B. Jobbágy and Jackson, 2000; Rumpel and Kögel-Knabner, 2011). Dies haben auch Studien aus den Schweizer Alpen, Island und Grönland gezeigt (Kolka-Jónsson, 2011; Sjögersten et al., 2011; Petrenko et al., 2016; Henkner et al., 2016). Je nach Bodentyp, Vegetation (z.B. tiefwurzelnde Grasarten) und regionalen Standorteigenschaften können beträchtliche Mengen an Kohlenstoff tiefer als 30 cm vorhanden sein (Jobbágy and Jackson, 2000; Tarnocai et al., 2009). So sind nach Tarnocai et al. (2009) 61 % des totalen Bodenkohlenstoffes in der zirkumpolaren Permafrostzone tiefer als 30 cm gespeichert. In tieferen Bodenschichten ist der SOC meist stabiler gebunden und weist aufgrund seiner räumlichen Trennung vor dem Edaphon eine längere Verweildauer im Boden auf (Chabbi et al., 2009; Trumbore, 2009; Schmidt et al., 2011; Rumpel and Kögel-Knabner, 2011).

Unter der Betrachtung der Beprobungstiefe hat somit in den drei Fallstudien die Bestimmung des Bodenkohlenstoffvorrates bis zu einer Tiefe von 30 cm höchstwahrscheinlich zu einer Unterschätzung des Kohlenstoffvorrates im Boden geführt, weil nicht der gesamte Kohlenstoffvorrat bis zum Ausgangsmaterial ermittelt worden ist (Sjögersten et al., 2011). Nach Nussbaum et al. (2012) werden in den Alpen (> 1200 m ü. M.) 69.5 t C ha^{-1} und 115.3 t C ha^{-1} in den ersten 30 cm und 100 cm gespeichert. Die natürlichen Standortvoraussetzungen des Unteralptals hätten eine Beprobung des Bodens tiefer als 30 cm jedoch nur bedingt ermöglicht. Denn bereits in den ersten 30 cm sind von den total 863 Bodenproben aufgrund des hohen Gehalts an Gesteinsschuttmaterial nur 506 Proben mit dem Stechzylinder entnommen worden.

Für die Kohlenstoffvorräte des Datensatzes „Aufforstung“ gilt, dass die Böden, durch Erosions- und Depositionsprozesse geprägt sind (Arnalds, 2015g). So können von der Erosion verschonte Andosolböden eine Bodenmächtigkeit von mehreren Metern erreichen und vertikal betrachtet aus mehreren übereinander liegenden Paläoböden bestehen (Helgason, 1899; Kolka-Jónsson, 2011). Die in der Vergangenheit rezenten Böden mit dem während ihrer Bildung gespeicherten organischen Kohlenstoff sind von Vulkanausbruchmaterial überdeckt worden und die Bodenbildung hat an der Geländeoberfläche neu begonnen, was zu einer Konservierung des Bodenkohlenstoffes geführt hat (Arnalds, 2015g). Die Fallstudie „Aufforstung“ mit der Abschätzung des Bodenkohlenstoffvorrates in den ersten 30 cm unterschätzt somit deutlich den maximalen Kohlenstoffvorrat der beprobten Böden. Hinsichtlich des Effekts der Aufforstung auf die zusätzliche Bodenkohlenstoffspeicherung ist die gewählte Beprobungstiefe von 30 cm als ausreichend betrachtet worden, weil *B. pubescens* hauptsächlich in den ersten 30 cm wurzelt (Hunziker et al. 2014). Die Studie hat zudem gezeigt, dass das Signal der Auswirkung der Aufforstung auf die SOC Veränderung durch bereits vorhandenen Kohlenstoff gestört wird (Kapitel 4).

Jandl et al. (2014) verlangt in allen Fällen die Beprobung des Auflagehorizontes, welcher als zusätzliches Kohlenstoffreservoir gilt (Aalde et al., 2006a). Studien zeigen, dass bei der Entstehung von Wald- oder Buschvegetation auf ehemaligen Grasflächen eine Streuauflage gebildet wird, weil die Bildungsrate der Auflage grösser als die Einarbeitungsrate in den mineralischen Boden ist (Hiltbrunner et al., 2013; Guidi et al., 2014a; Bühlmann et al., 2016). Während der Verbuschung durch *A. viridis* ist eine Zunahme des Kohlenstoffvorrates im Auflagehorizont von 1.0-1.5 t C ha^{-1} gemessen worden (Bühlmann et al., 2016). Der produktive Lebensraum *Alnenion viridis* (Delarze et al., 2015) mit einer jährlichen Blattbiomasseproduktion der Grünerle von 3.8 t ha^{-1} (Wiedmer and Senn-Irlet, 2006) bildet bei einem Bestandesalter von 90 Jahren im Herbst einen Laubauflagehorizont dessen Bestimmung es in Zukunft zu berücksichtigen gilt (Abbildung 25). Nebst der Untersuchung des Kohlenstoffvorrates im mineralischen Boden und im Auflagehorizont müssten zudem die Vorräte in der Buschbiomasse und in der Krautschicht ermittelt werden, um ein holistischeres Bild der Kohlenstoffveränderung durch die Grünerlenverbuschung zu erhalten (Bühlmann et al., 2016).

Abbildung 25: Situation nach dem Laubabfall und Zusammenfallen des Unterwuchses (hier v.a. *Oreopteris limbosperma*) im *Alnenion viridis*, bei dem vor 90 Jahren das Grünerlenwachstum begonnen hat. Im Hintergrund befinden sich Gebäude der Ortschaft Andermatt. Aufgenommen: M. Hunziker, am 19. Oktober 2011.

Beprobungsschichten und Beprobungswerkzeug

Die Beprobung des Bodens hat mit einem Stechzylinder mit einem Volumen von 100 cm^3 (Eijkelkamp Soil & Water, Giesbeek) stattgefunden. Die Verwendung des Stechzylinders anstelle von Erdbohrern hat den Vorteil, dass der Boden in der Vertikalen aufgelöster beprobt werden kann und die Kohlenstoffeigenschaften differenzierter betrachtet werden können. Dies ist bei Vegetationsveränderungen verbunden mit einer Änderung der Verfügbarkeit an ober- und unterirdischem organischen Material essentiell, um eine Aussage über Änderung der räumlichen SOC Kohlenstoffvorrates machen zu können. Demnach ist in diesen beiden Bereichen das Level 3 von maximal 4 erreicht worden (Tabelle 16) (Jandl et al., 2014).

Dank der differenzierten Auflösung des Kohlenstoffverhaltens in der Vertikalen (0-5 cm, 5-10 cm, 10-20 cm und 20-30 cm) in Kombination mit der physikalischen Bodenfraktionierung hat die Studie „Aufforstung" zeigen können, dass bei der Inventarisierung des Bodenkohlenstoffes bei aufgeforsteten Flächen mit Kohlenstoff zu rechnen ist, welcher nicht von der aufkommenden Biomasse stammt (Kapitel 4). Dies wäre bei der Bestimmung des Bodenkohlenstoffvorrates (0-30 cm) auf Basis einer Mischprobe (0-30 cm), welche mit Hilfe eines Stechbohrers gezogen worden wäre, nicht ersichtlich gewesen. Zudem hat die vertikale Auflösung gezeigt, dass beim Einwachsen von *A. viridis* auf aufgelassenen Alpweiden die ausgeprägtesten und signifikanten Veränderungen im Beprobungshorizont „0-5 cm" stattfinden (Kapitel 3) und das übliche Verhalten in Form einer Abnahme des Kohlenstoffvorrates mit Zunahme der Tiefe bereits in den vier Beprobungshorizonten bereits feststellbar und daher mit anderen Studien vergleichbar ist (Leifeld et al., 2009; Budge et al., 2011; Guidi et al., 2014a).

6.1.2 Probeumfang im Feld und Replikate in der statistischen Analyse

Pro Standort, mit Ausnahme der Standorte in der Fallstudie „Klimaerwärmung“, welche der Tundravegetation zugewiesen worden sind, ist der Boden jeder der vier Beprobungsschichten durch fünf Proben repräsentiert worden (Level 4). Damit ist die repräsentative Abbildung der Bodeneigenschaften des Standortes gewährleistet gewesen. Der Arbeitsaufwand, der durch den Einbezug der Bodenfraktionierung mittels Dichtetrennung im Vergleich zur Herleitung des allgemeinen SOC Vorrates viel höher ausgefallen ist, hat nach einer Reduktion des Probeumfangs verlangt. Daher sind die fünf Replikate pro Beprobungsschicht eines Standortes im Feld („Aufforstung“) oder im Labor („Landaufgabe“ und „Klimaerwärmung“) zu gleichen Massenteilen zu einer Mischprobe zusammengeführt worden (Level 1). Mit der Mischung des Bodenmaterials hat sich auch die räumliche Variation verringert, was nicht Gegenstand der Untersuchungen gewesen ist. In Landschaften mit rauem Relief (Hoffmann et al., 2014b) oder bei Störungen der Bodenverhältnisse durch Erosions- und Akkumulationsprozesse, wie in Island, ist jedoch mit einer erhöhten räumlichen Variation zu rechen und sollte in solchen Fällen bedacht werden.

Das Weiterarbeiten mit Mischproben hat zur Folge gehabt, dass die Anzahl Replikate pro Strata und Bodenschicht für „Landaufgabe“ N=9, „Aufforstung“ N=3 betragen hat und bei „Klimaerwärmung“ zwischen N=1 und N=7 variiert hat. Aufgrund der tiefen Zahl an Replikaten und damit verbunden das Nichterreichen von normalverteilten Datensätzen sind deskriptive Statistikmethoden und, wenn es der Probeumfang erlaubt hat („Landaufgabe“), nichtparametrische Tests zur Identifizierung von Unterschieden zwischen den Strata und/oder Bodenschichten verwendet worden.

6.1.3 Fehlerquellen bei der Abschätzung des Kohlenstoffvorrates im Bodensystem

Poeplau et al. (2017) hat vier verschiedene Formeln zur Abschätzung des Bodenkohlenstoffvorrates untersucht und dabei verschiedene Herleitungen der Bodendichte sowie die Auswirkungen des Steingehaltes auf den Bodenkohlenstoffvorrat evaluiert. Die Arbeit zeigt deutlich auf, dass drei Methoden den Kohlenstoffvorrat im Boden teilweise bis zu über 100% überschätzen. Die Autoren schlagen daher vor, dass in Zukunft für die Abschätzung des Kohlenstoffvorrates der Anteil des Feinbodens und wenn möglich auch der Steingehalt miteinbezogen werden. Dies ist übereinstimmend mit „Level 4“ von Jandl et al. (2014) beim Thema „Bodendichte“. In der vorliegenden Arbeit hat das Volumen des Beprobungswerkzeuges 100 cm^3 betragen. Für die Berechnung des Bodenkohlenstoffvorrates ist die Bodendichte korrigiert worden. Dabei ist das Feinbodenmaterial vom Skelettgehalt (> 2 mm), welcher aus Wurzelbiomasse und/oder mineralischer Substanz bestehen kann, separiert worden. Die Masse sowie das Volumen des Skelettgehaltes sind mittels Messung der Masse und der

Volumenverdrängung in Wasser bestimmt worden (Level 4). Für die Bestimmung der Dichte des Feinbodenmaterials, welche als Quotient von Masse zu Volumen definiert ist, sind vorgängig sowohl Masse als auch Volumen des Feinbodenmaterials durch Subtraktion der Grössen des Skelettgehaltes von jenen der Gesamtbodenprobe ermittelt worden.

Grundsätzlich lassen die durchgeführten Analysen nur eine Aussage über den Bodenkohlenstoffvorrat von Bodenmaterial zu, das mittels Stechzylinder hat beprobt werden können. Das Steinmaterial (> 100 cm^3) ist jedoch in allen Fallstudien im beprobten Boden (0-30 cm) vorhanden gewesen. Die Quantität des Steinmaterials wird dabei subjektiv als „Landaufgabe“ >> Fallstudie „Aufforstung“ > Fallstudie „Klimaerwärmung“ beurteilt. Weiter ist der Anteil an Felsschutt (FAO 2006) nicht in die Abschätzung des Bodenkohlenstoffvorrates eingeflossen (Level 1). Diese beiden mineralischen Feststoffe im Boden können aber aufgrund von geomorphologischen Prozessen und lithologischen Verhältnissen charakteristisch für Landschaften sein. Die Nichtberücksichtigung des Anteils an Felsschutt und des Steingehaltes (> 100 cm^3) im Boden, welcher zudem gemäss Fallstudie „Landaufgabe“ mit der Tiefe zugenommen hat, führt somit zu einer Überschätzung des eigentlichen Bodenkohlenstoffgehaltes in der beprobten Tiefe von 30 cm. Denn durch die Nichtberücksichtigung wird angenommen, dass die beprobte Bodenmächtigkeit, in diesen Fällen 0-30 cm, nur aus Material <100 cm^3 besteht und durch das Probematerial repräsentiert wird.

Ein drittes Kohlenstoffreservoir nebst jenem des Auflagehorizontes und des mineralischen Bodens bildet die Wurzelbiomasse. Definitionsgemäss wird das lebendige Wurzelmaterial (> 2 mm) der Biomasse zugewiesen (Vogt et al., 1996). Die Wurzelbiomasse wird mittels allometrischen Funktionen, die unter aufwändiger Feldarbeit hergeleitet werden und daher nicht für alle Pflanzenarten verfügbar sind, oder mittels „root:shoot ratios“ abgeschätzt. Im Rahmen des CarbBirch Projekts in Island hat Hunziker et al. (2014) für *B. pubescens* Ehrh. *ssp. czerepanovii* allometrische Funktionen für die Wurzelbiomasse (> 2 mm) hergeleitet und dazu dieselben Testflächen wie in der Fallstudie „Aufforstung“ genutzt. Die Abschätzungen des Kohlenstoffvorrates in der Wurzelbiomasse und im mineralischen Boden können somit in Verbindung gebracht werden. Der Kohlenstoff in der Wurzelbiomasse wird für Birch50 und Birchnat auf 21.3 und 9.2 t C ha^{-1} (0-30 cm) geschätzt. Der Bodenkohlenstoffvorrat beträgt für dieselben Standorte 46.5 und 59.4 t C ha^{-1} (0-30 cm) (Abbildung 16). Das unterschiedliche Verhältnis zwischen Kohlenstoffvorräten in der Wurzelbiomasse und im mineralischen Boden ist auf die unterschiedlichen Entwicklungsstadien, in denen sich Birch50 und Birchnat befinden, zurückzuführen (Smith et al., 1997). Birch50 hat die Selbstverjüngungsphase noch nicht abgeschlossen. Folglich wird bei Birch50 der Wurzelbiomassevorrat und damit verbunden auch der Kohlenstoffvorrat in den Wurzeln in Zukunft wieder abnehmen und sich Birchnat angleichen. Hingegen ist bei Birch50 weiterhin ein SOC Senkenpotential im Boden im Vergleich zu Birchnat (Δ 13 t C ha^{-1}) vorhanden. Damit kann gezeigt werden, dass sich das Kohlenstoffvorrats-verhältnis zwischen Wurzelbiomasse und mineralischem Boden während der Etablierung eines neuen Vegetationstyps verändert. Dies sollte bei der Wahl von Probestandorten für Kohlenstoffinventare und der Schlussfolgerung auf Kohlenstoffvorräte und –flüsse des entstehenden Ökosystems berücksichtigt werden.

Eine weitere Berücksichtigung der Wurzelbiomasse hat nur in der Fallstudie „Klimaerwärmung“ im Zusammenhang mit der Abschätzung der Gesamtbiomasse der Buschvegetation nach Hunziker et al. (2014) stattgefunden, welche aber nicht in die Abschätzung des Kohlenstoffs im Boden eingeflossen ist (Level 1). Das Bodensystem ganzheitlich betrachtet ist der gesamte Kohlenstoffvorrat im Boden, welcher den Auflagehorizont, den mineralischen Boden und die Wurzelbiomasse beinhaltet, durch die Aufforstung oder die Verbuschung infolge von Klimaerwärmung oder Landaufgabe noch höher anzunehmen als die drei Studien mit der Abschätzung des mineralischen Bodenkohlenstoffvorrates hervorgebracht haben.

6.1.4 Abschätzung des Kohlenstoffvorrates in den SOC-Fraktionen

Nach der Charakterisierung des Bodenkohlenstoffes mit Hilfe von Fraktionierungsverfahren und der Ermittlung der Kohlenstoffkonzentrationen [mg C g^{-1} Boden] kann es anschliessend von Interesse sein, den Kohlenstoffvorrat [t C ha^{-1}] in den einzelnen Fraktionen auf die Fläche hochzurechnen. Dies ist von besonderer Bedeutung, wenn sich Bodensysteme infolge eines Landbedeckungswandels ändern und der labile Anteil des Bodenkohlenstoffes zunimmt, was weniger im Interesse einer langfristigen Speicherung ist. Doch die Reservoire der SOC Fraktionen verhalten sich während eines Vegetationswandels dynamisch (Kapitel 3-5), weshalb sich auch der Kohlenstoffvorrat in diesen Reservoiren mit dem Wandel verändert. Der Kohlenstoffvorrat im Boden (SOC stock; mg cm^{-2}) lässt sich durch Multiplikation der Kohlenstoffkonzentration (SOC; mg g^{-1} Boden) mit der Feinbodendichte (g cm^{-3}) und der Beprobungstiefe (cm) berechnen (Kapitel 3-5). Die Anwendung der Gleichung ist für die Abschätzung der Kohlenstoffvorräte in den SOC Fraktionen nicht möglich, weil die Dichte der Materialien in den Fraktionen nicht bekannt und ermittelt worden ist.

Guidi et al. (2014a) hat versucht den Kohlenstoffvorrat in der POM-, HF- und S+C-Fraktion abgeschätzt. Dabei entsprechen die Vorräte in den Fraktionen dem relativen Verhältnis der Kohlenstoffkonzentrationen der einzelnen Fraktionen, welche mit dem Gesamtkohlenstoff verrechnet worden sind (persönliche Mitteilung). Auf Basis dieser Herangehensweise sind auch die Kohlenstoffvorrate der Fraktionen in dieser Arbeit berechnet worden, weil keine zufriedenstellendere Alternativmethode gefunden werden konnte.

Hinsichtlich der Resultatqualität der Kohlenstoffvorräte in den einzelnen Fraktionen ist davon auszugehen, dass die Nichtberücksichtigung der Materialdichten in den Fraktionen einen erheblichen systematischen Fehler verursacht, welcher gegenwärtig in den Vorräten der Fraktionen enthalten ist. Denn das Material der POM-Fraktion weist aufgrund des gewählten Fraktionierungsmittels eine Dichte von < 1.8 g cm^{-3} (Kapitel 4, Kapitel 5) resp. < 2.0 g cm^{-3}

(Kapitel 3) auf. Hingegen wird das mineralische Material, aus dem vorwiegend die HF- und S+C-Fraktionen bestehen, gewöhnlich mit einer Dichte von ca. 2.6 g cm^{-3} angegeben. An dieser Stelle sollte bedacht werden, dass die Porosität des Materials und damit der Lufteinschluss die Dichte des Bodenprobematerials herabsetzt. Aus diesem Grund betragen die mittleren Dichten des Feinbodenmaterials (unfraktioniert), welches aus organischem und mineralischem Material besteht, 0.58 („Landaufgabe"), 0.71 („Aufforstung) und 0.77 g cm^{-3} („Klimawandel"). Aufgrund der unterschiedlichen Dichten des Materials in den Fraktionen und der unfraktionierten Bodenprobe (< 2 mm) lässt sich schliessen, dass mit der angewendeten Methode, welche im Ursprung die Dichte des Feinbodenmaterials verwendet, der Kohlenstoffvorrat in der POM-Fraktion überschätzt und jener der S+C-Fraktion unterschätzt wird. Daher wird an dieser Stelle geraten, dass in Zukunft vorgängig an die Berechnung der Kohlenstoffvorräte in den einzelnen Fraktionen die Dichten des jeweiligen Materials ermittelt werden.

6.1.5 Verwendete Methode zur Bestimmung der Qualität des Bodenkohlenstoffes

Die Störung des Bodenkohlenstoffgleichgewichts in Form einer Vegetationsveränderung hat eine Veränderung des Bodenkohlenstoffes zur Folge (Six et al., 2002b). Weil die organische Bodensubstanz als heterogen betrachtet wird (Schmidt et al. 2011), verändert sich bei einem Landbedeckungswandel auch die Zusammensetzung der organischen Substanz. Die vorliegende Arbeit hat deshalb nebst der Veränderung der Kohlenstoffvorräte auch die Auswirkung der Vegetationsveränderung auf die Qualität des Bodenkohlenstoffes untersucht. Wie von Jandl et al. (2014) empfohlen, sind die Proben einer physikalischen Fraktionierung nach Grösse und Dichte der Bodenmaterialien unterzogen worden (Level 4). Physikalische Fraktionierungen des Bodens eignen sich zur Beschreibung und Quantifizierung der qualitativen Kohlenstoffveränderungen, die durch Vegetationsveränderungen verursacht werden (Elliott and Cambardella, 1991; Six et al., 1998; Poeplau and Don, 2013). Bei allen drei Fallstudien ist das Fraktionierungsverfahren von Zimmermann et al. (2007b) angewendet worden. Das Flussdiagramm in Abbildung 26 zeigt die einzelnen analytischen und rechnerischen Teilschritte auf, um die Konzentrationen und Vorräte des Kohlenstoffs in den einzelnen Fraktionen und im gesamten Boden (0-30 cm) zu eruieren.

Das Verfahren teilt den Bodenkohlenstoff in vier verschiedenene funktionellen Kohlenstoffreservoirs, welche im Folgenden aufgrund ihrer Erzeugung nach Zimmermann et al. (2007b) und Eigenschaften bezüglich der Kohlenstoffstabilität definierend beschrieben werden:

- POM-Fraktion

  Nach der Dispergierung (22 J ml^{-1}) und der Nasssiebung (63 µm) kann die organische Substanz, welche grösser als 63 Mikrometer und nicht mit der mineralischen Phase aggregiert ist, mittels Dichtetrennung von der Fraktion, welche aus organischer sowie mineralischer Substanz der Sandfraktion besteht, separiert werden. Die POM-Fraktion beinhaltet somit organisches Material, das sich frei und ungebunden in der Bodenmatrix befindet, sowie ursprünglich okkludiertes Material, das durch die Dispergierung aus aufgebrochenen Aggregaten frei geworden ist (von Lützow et al., 2008). Die Zusammensetzung besteht aus unterschiedlich stark zersetzten Pflanzenresten (oberirdische Streu und Wurzelmaterial), Samenmaterial sowie Material der Bodenfauna und –flora (Christensen, 2001). Im Hinblick auf die Verweilzeit des Materials im Boden handelt es sich um labilen Kohlenstoff, weil er für Bodenorganismen teils frei zugänglich ist und deshalb innerhalb von Monaten und Jahren umgesetzt wird (Sollins et al., 1996).

- HF-Fraktion

  Eine weitere Kohlenstoff-Fraktion, welche bei der Dichtetrennung gewonnen wird, ist das organische Material, das mit der mineralischen Phase verbunden ist. Dabei handelt es sich um primäre oder sekundäre organo-mineralische Komplexe der Sandgrösse (> 63 µm) (Christensen, 2001). Trotz der vorgängigen Ultraschallbehandlung kann der mineralische Teil dieser Komplexe nebst Einzelsandkörnern auch aus Aggregaten bestehen, die aus Partikeln der Sand-, Schluff- oder Tonfraktion aufgebaut sind und durch biologische Aggregierung oder Komplexierung mit Ionen ($Na^+ < K^+ < Ca^{2+} < Al^{3+} < Fe^{3+}$) entstanden sind (Sollins et al., 1996). Die Verweildauer des darin gebundenen Kohlenstoffs kann einige Jahre bis Jahrzehnte betragen (Sollins et al., 1996).

- S+C-Fraktion

  Das Bodenmaterial, welches bei der Nasssiebung kleiner als 63 µm ist, wird nach Zimmermann et al. (2007b) der Schluff- und Tonfraktion zugewiesen. Mikroaggregierung (20-63 µm) und die Bildung von Mikrostrukturen mit Tonmineralien (< 20 µm) spielen bei der Stabilisierung eine Rolle (von Lützow et al., 2008). Die Verweildauer des Kohlenstoffs dieser Fraktion im Boden kann Jahrzehnte bis mehr als 100 Jahre betragen (Sollins et al., 1996). Die Kohlenstofffraktion wird somit als S+C-Fraktion definiert. Je nach Bodenklassifikation wird die Grenze zwischen der Sand- und Schlufffraktion bei 63 µm (bspw. Deutschland, WRB), 74 µm (USCS) oder 50 µm (bsw. Frankreich, Russland, USA, Schweiz) gezogen (Guidelines for Classification and Diagnostics of Soils, 1967; Ad-hoc-AG Boden, 2005; Baize et al., 2009; BGS, 2010; Soil Survey Staff, 2014; IUSS Working Group WRB, 2014), was bei Vergleichen zwischen der Kohlenstoffkonzentrationen von S+C Fraktionen anderer Studien berücksichtigt werden muss.

- DOC-Fraktion

  Nebst der Bodenatmung des Edaphons kann der Kohlenstoff über das Bodenwasser aus dem System ausgetragen werden (Vesterdal et al., 2013). Zur Bestimmung des

Gehalts an gelöster, organischer Substanz in der Bodenprobe, wird eine Probe der Suspension, bestehend aus Schluff- und Tonmaterial sowie destilliertem Wasser, filtriert und anschliessend auf ihren Gehalt an gelöstem Kohlenstoff analysiert. Diese Bodenkohlenstoffform wird der labilen Fraktion zugeordnet.

Die Meinung des Autors ist, dass der Kohlenstoff, welcher in der DOC-Fraktion (< 45 µm) gemessen wird, lediglich als Indikator genutzt werden kann, um abschätzen zu können, wieviel Kohlenstoff in der Bodenprobe potentiell mobilisiert wird und vom Bodensystem ins aquatische System übertreten kann. Durch die Verwendung von diesem Indikator als Analysemethode für den gesamten Datensatz, lassen sich die Böden unterschiedlichen Verbuschungsgrad trotzdem vergleichen und der Einfluss der Vegetationsveränderung auf die Labilität des Bodenkohlenstoffes kann abgeschätzt werden. Genauere Messungen des DOC-Gehaltes im Boden, was bei der Verbuschung durch die Grünerle von Interesse sein kann (Abbildung 13), müssten mittels Feldmessungen durchgeführt werden (Hagedorn et al., 2000).

Bei der Fraktionierung des Bodenkohlenstoffes ist auf die chemische Separierung, die Nassoxidation mit Natriumhypochlorit (NaOCl), verzichtet worden. In einer Voranalyse ist mit einem Teildatensatz (N=42) aus dem Projekt „Aufforstung“ die Durchführbarkeit der Methode getestet worden. Die Evaluation inkl. Literaturstudium hat jedoch ergeben, dass der Aufwand an Ressourcen (Zeit und Chemikalien) nicht im Verhältnis zur Aussagekraft der Resultate der sogenannten rSOC-Fraktion und zum Volumen an Chemikalienabfall bei der Anwendung an 333 Proben steht. rSOC steht für resistentes SOC, das biochemisch rekalzitrant ist (Zimmermann et al., 2007a). Zudem wird in der jüngeren Vergangenheit die Nassoxidation hinsichtlich ihrem Ziel, den chemisch rekalzitranten Kohlenstoff bestimmen zu können, in Frage gestellt (Jagadamma et al., 2010; Lutfalla et al., 2014) Weiter scheint die Rekalzitranz vor allem in den ersten Zersetzungstadien der organischen Substanz von Bedeutung zu sein (Lützow et al., 2006; Schmidt et al., 2011). Dieses kaum zersetzte Material wird in der vorliegenden Arbeit jedoch bereits der POM-Fraktion zugeordnet. Weiter wird in dieser Arbeit angenommen, dass die S+C-Fraktion bereits biologisch bearbeiteter Kohlenstoff enthält (Jandl et al., 2014), welcher vorwiegend durch abiotische Aggregierungsvorgänge mit der Schluff- und Tonmaterialien und somit in organo-mineralischen Komplexen stabilisiert ist (von Lützow et al., 2008).

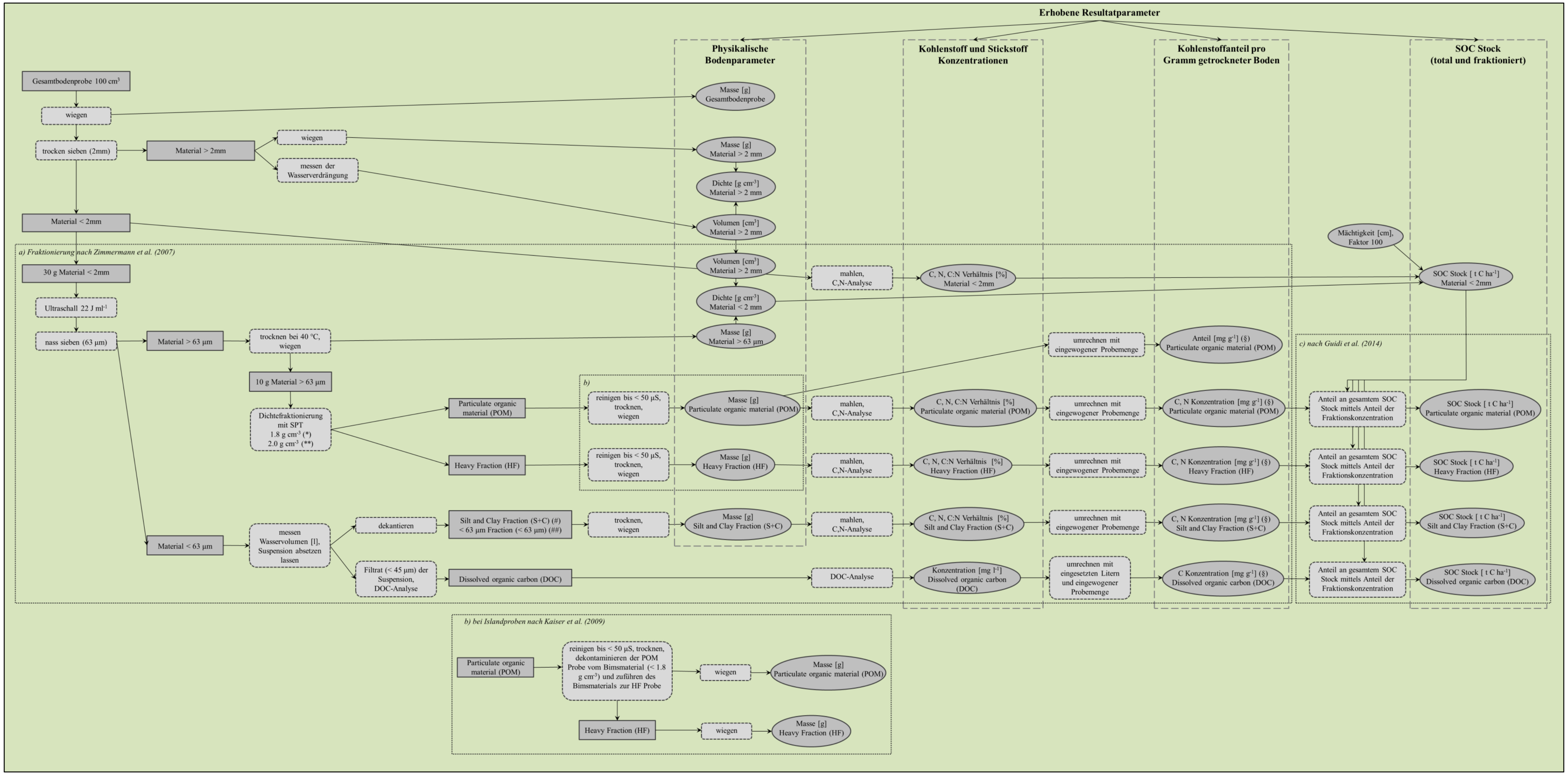


Abbildung 26: Flussdiagramm des Programms der Bodenfraktionierung im Labor (links und mittig) und der anschliessenden Herleitung der Kohlenstoffkonzentrationen und des –vorräte im Gesamtboden und in den einzelnen Fraktionen (rechts), wobei unterschieden wird zwischen: Bodenmaterial (eckiger, durchgezogener Rahmen und dunkelgrauer Hintergrund), Methode/Prozess (abgerundeter, gestrichelter Rahmen und hellgrauer Hintergrund) und erhobene Resultatparameter (ovaler, durchgezogener Rahmen und dunkelgrauer Hintergrund). Das Analyseschema beinhaltet die angepasste Fraktionierungsmethode (ohne NaOCl Oxidation) von Zimmermann et al. (2007b) (a), die Dekontaminierung des POM Materials vom mineralischen Bims-Material bei den Isländischen Proben mittels elektrostatischer Anziehung nach Kaiser et al. (2009) (b) und die Berechnung der SOC Vorräte in den einzelnen Fraktionen (POM, HF, S+C und DOC) nach Guidi et al. (2014a) (c).

6.1.6 Abschliessende Beurteilung

Laut Jandl et al. (2014) bieten sich alle in ihrer Studie aufgeführten Punkte unabhängig ihrer Einstufung zur Abschätzung des Bodenkohlenstoffvorrates an (Tabelle 16). Jedoch raten die Autoren Bodenkohlenstoffveränderungen (z.B. beim Landbedeckungswandel) in quantitativer und qualitativer Form zu untersuchen, wobei nur Methoden ab Level 4 verwendet werden sollten. Diese Forderung wird hinsichtlich der Generierung von künftigen qualitativ hochwertigen Datensätzen auf Basis der Wahl der Methoden (Tabelle 16) geteilt. Diese sehr wertvollen Datensätze können später für räumliche Modellierungen des Bodenkohlenstoffvorrates auf regionaler oder globaler Ebene verwendet werden. Doch es wird davon ausgegangen, dass es schwierig ist, diese Form der Bodenkohlenstoffinventarisierung inkl. der qualitativen Beurteilung des Kohlenstoffs aufgrund von fehlender Infrastruktur (z.B. Laboreinrichtungen) oder Ressourcen wie Arbeitskräfte, Zeit und Geld durchführen zu können. Daher wird es in vielen Fällen zu einer Priorisierung der Themen, welche von Jandl et al. (2014) aufgeführt werden, kommen. Auf Basis der Resultate (Kapitel 3 bis 5) und der darauf aufbauenden Diskussion (Kapitel 6) sind nach Einschätzung des Autors die Themen „Proben pro Standort“, „Beprobungswerkzeug“, „Beprobungsschichten“, „Beprobungstiefe“, „Bodendichte“, „Steingehalt“, „Kohlenstoffgehalt“ und „SOC-Qualität“ zu priorisieren.

Die von Jandl et al. (2014) publizierte Tabelle (Tabelle 16) lässt eine Bewertung der Datensätze und der angewendeten Methoden zu. Dem Evaluationsbogen nach können maximal 56 Punkte vergeben werden Die vorliegenden Datensätze der Fallstudien erhalten 37 („Landaufgabe“) und 35 („Aufforstung“ und „Klimaerwärmung“) Punkte (Tabelle 16). Mehr Punkte hätten erziehlt werden können, wenn bei der Probenahmen im Feld die für Kohlenstoffinventare relevanten Faktoren „Wurzelgehalt“, „Totholz“, „Fels“ und „Blöcke“ berücksichtigt und der Auflagehorizont mitbeprobt worden wären. Im Labor hätten die Proben zudem einer Textur- und IR-Spektroskopieanalyse unterzogen werden können. Jedoch fliessen die erwähnten mitbestimmenden Faktoren nicht direkt in die Formel zur Berechnung des Kohlenstoffvorrates ein (Aalde et al., 2006a) und dienen daher eher zur Beschreibung der Untersuchungsgebiete in Bezug auf die Evaluierung der Kohlenstoffvorräte. Für die Mitberücksichtigung dieser Faktoren müsste eine neue Standardmethode zur Ermittlung des Bodenkohlenstoffvorrates entwickelt und anerkannt werden.

Tabelle 16: Beurteilung der Datensätze der drei Fallstudien anhand der Level des Evaluationsbogens nach Jandl et al. (2014)

Themen	Fallstudie „Landaufgabe“		Fallstudie „Aufforstung“		Fallstudie „Klimaerwärmung“	
Beprobungsstrategie	Stratified random sampling	■■■	Stratified random sampling	■■■	Stratified random sampling	■■■
Proben pro Standort	N = 5	■■■■	N = 5	■■■■	N = 5, N = 1 (Refs)	■■■■
Beprobungswerkzeug	Stechzylinder (100 cm^3)	■■■	Stechzylinder (100 cm^3)	■■■	Stechzylinder (100 cm^3), Erdbohrer (Refs)	■■■
Tiefe	Mineralischer Boden: 30 cm	■■	Mineralischer Boden: 30 cm	■■	Mineralischer Boden: 30 cm	■■
Beprobungsschichten	0-5, 5-10, 10-20, 20-30 cm	■■■	0-5, 5-10, 10-20, 20-30 cm	■■■	0-5, 5-10, 10-20, 20-30 cm	■■■
Bodendichte	Messung im Labor	■■■■	Messung im Labor	■■■■	Messung im Labor	■■■■
Steingehalt	Volumetrische Bestimmung	■■■■	Volumetrische Bestimmung	■■■■	Volumetrische Bestimmung	■■■■
Wurzeln	Keine Bestimmung	■	Keine Bestimmung	■	Keine Bestimmung	■
Fels, Blöcke, Totholz	Keine Bestimmung	■	Keine Bestimmung	■	Keine Bestimmung	■
Mischung	1 Mischprobe aus 5 Proben	■	1 Mischprobe aus 5 Proben	■	1 Mischprobe aus 5 Proben bei Birken	■
Kohlenstoffgehalt	Verbrennungsmethode	■■■	Verbrennungsmethode	■■■	Verbrennungsmethode	■■■
Fraktionierung	Grössen- und Dichtetrennung	■■■■	Grössen- und Dichtetrennung	■■■■	Grössen- und Dichtetrennung	■■■■
IR Spektroskopie	Keine Analyse	■	Keine Analyse	■	Keine Analyse	■
Texturanalyse	Caviezel et al. (2014)	■■■	Keine Analyse	■	Keine Analyse	■

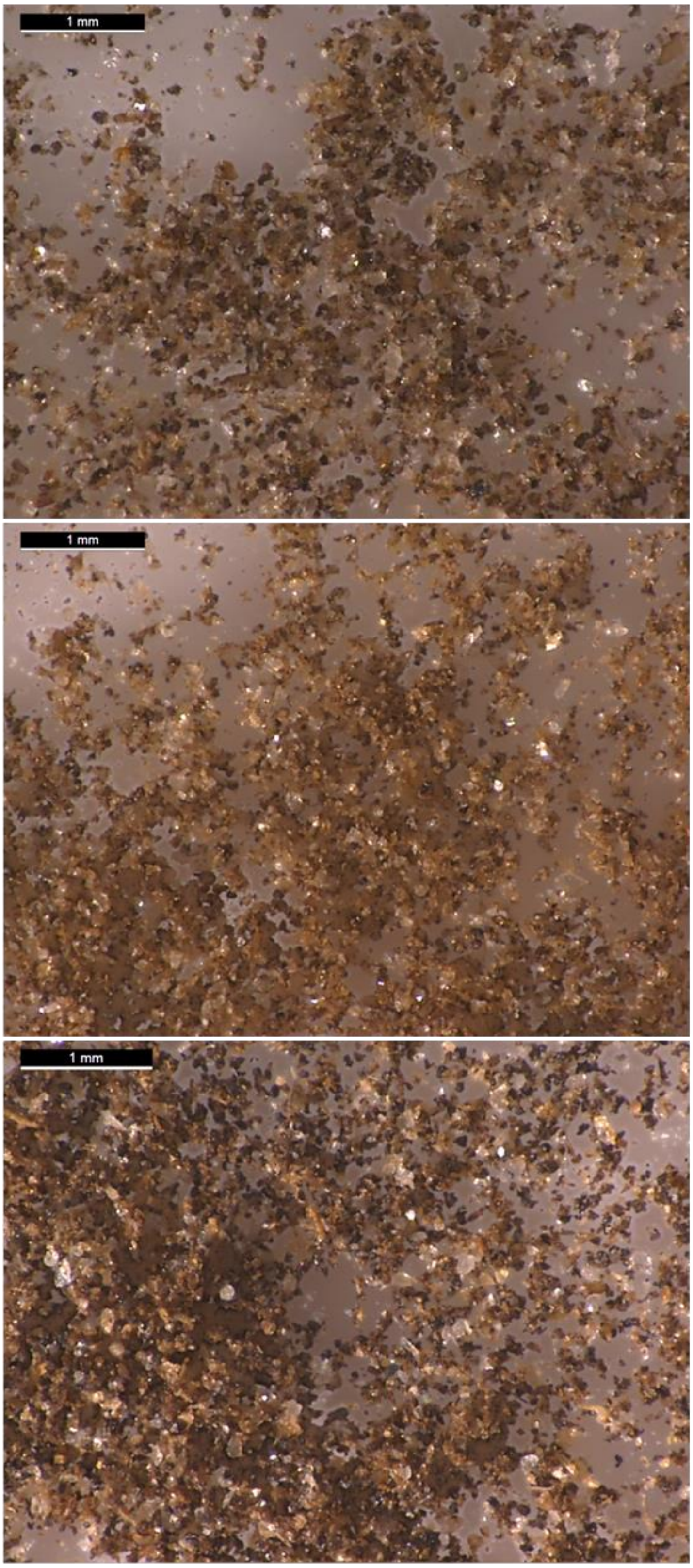
Level 1:	■
Level 2:	■■
Level 3:	■■■
Level 4:	■■■■

6.2 Einschätzungen zur Veränderung des Kohlenstoffs in der „Schluff- und Ton“-Fraktion

Die Resultate der Fallstudien „Landaufgabe“ und „Aufforstung“ weisen eine Zunahme der Kohlenstoffkonzentration [mg g^{-1} Boden] in der S+C Fraktion während der Etablierung der neuen Vegetationsbedeckung auf (Abbildung 12, Abbildung 17). Bei der Verbuschung durch die Grünerle verhält sich die Konzentration (Medianwert) in den obersten 5 cm für Control, GA15, GA25, GA40 und GA90 wie folgt: 18, 17, 20, 23 und 51 [mg g^{-1} Boden]. Dies entspricht einer prozentualen Zunahme der Konzentration zwischen einer Alpweide und einem 90 jährigen Grünerlenstandort von 183 %. Diese Zunahme ist indirekt auch beim visuellen Vergleich des Probematerials der S+C-Fraktion aufgrund der dunkleren Farbe des Materials der GA90 Proben ersichtlich gewesen. In den obersten 30 cm des Bodens nimmt bei der Aufforstung in Island mit *B. pubescens* Ehrh. die Kohlenstoffkonzentration in der S+C Fraktion von 30 mg C g^{-1} Boden (Birch15) auf 46 mg C g^{-1} Boden (Birch50) zu. Degradierte Böden weisen jedoch 57 mg C g^{-1} Boden auf. Während den Arbeiten haben sich daher folgende Fragen gestellt: Findet eine Interaktion der zusätzlichen organischen Substanz mit der mineralischen Schluff- und Tonphase statt? Und wird der darin enthaltene Kohlenstoff stabilisiert oder liegt das Material eher ungeschützt und kaum stabilisiert in der S+C-Fraktion vor? Es wurde versucht, die offenen Fragen mittels Mikroskop-Aufnahmen oder spektroskopischen Aufnahmen im visuellen und nahen Infrarot-Wellenlängenbereich zu beantworten. Die Ergebnisse werden in den folgenden beiden Kapiteln vorgestellt.

6.2.1 Beurteilung mittels Beschreibung von Mikroskop-Aufnahmen

Es ist zufällig ausgewähltes Probematerial der S+C-Fraktion von den obersten 5 cm der Verbuschungskategorien „Control“, „GA40“ und „GA90“ der Fallstudie „Landaufgabe“ verwendet worden. Beim Gerät handelt es sich um ein digitales Mikroskop vom Modell „Leica DMS 1000“ (Leica Microsystems CMS GmbH, Wetzlar). Das Material ist mit einer 4-fachen Vergrösserung und der Leica Application Suite Software (LAS V 4.9) aufgenommen worden. Die gleichgehaltenen Grundeinstellungen sind: Vergrösserungsstufe 4.0, Licht: von der linken Seite und von oben, gläserne Petrischale auf mattem, weissem Papier. Die Analyse bringt zum Vorschein, dass nach der Nasssiebung (< 63 µm) und dem Trocknen bei 40 °C mehr dunkles Material mit einem hohen Anteil von organischer Substanz bei „GA90“ als bei „Control“ oder „GA40“ vorhanden ist (Abbildung 27). Nebst mehr dunklen Aggregatkörnern sind auch mehr Wurzelhaare bei „GA90“ gefunden worden. Ein verstärktes Auftreten von Aggregaten mit schluffigen Mineralpartikeln bei „GA90“ kann nicht festgestellt werden. Die Methode kommt somit zum Schluss, dass das organische Material sehr wahrscheinlich mit sich selbst oder mit der Methode nicht ermittelbaren Tonmineralien verkittet ist und Aggregatstrukturen bildet. Die Zunahme der Kohlenstoffkonzentration in der S+C-Fraktion kann mit höchster Wahrscheinlichkeit nicht mit einem höheren Anteil an stabilisiertem Kohlenstoff in dieser Fraktion in Verbindung gebracht werden.



Standort: Control_0-5 cm

C-Konzentration: 11.6 % C

Standort: GA40_0-5 cm

C-Konzentration: 14.2 % C

Standort: GA90_0-5 cm

C-Konzentration: 25.2 % C

Abbildung 27: Mikroskopische Darstellung einer Auswahl der “S+C” Fraktion des Datensatzes „Landaufgabe“. Die Aufnahmen sind mit einem digitalen Mikroskop vom Modell „Leica DMS 1000“ (Leica Microsystems CMS GmbH, Wetzlar) und der Leica Application Suite Software (LAS V 4.9)

6.2.2 Beurteilung mittels Kombination von Spektraldaten und Cluster-Analyse

Bodenkohlenstoff kann mittels Spektralanalysen charakterisiert werden (Stenberg et al., 2010). Folglich lässt sich auch ein Signal in den spektralen Eigenschaften finden, wenn sich der Kohlenstoff aufgrund von Landbedeckungsänderungen verändert. Das Feinbodenmaterial der 84 Proben aus der Fallstudie „Aufforstung“ ist an der Universität Osnabrück von Dr. Thomas Jarmer der Spektralanalyse (vis-NIR Wellenlängenbereich) mit einem ASD Field Spec II Spektroradiometer (Analytical Spectral Devices) unterzogen worden. Die Aufnahmen sind in einem Wellenlängenbereich zwischen 350 und 2500 nm (1 nm Intervalle) mit einer 1000 W Quartz-Halogen Lampe (Lowel Pro-Lite P-10) im Abstand von 30 cm und einem Beleuchtungswinkel von 30 ° durchgeführt worden. Die Korrektur der Spektradaten hat mit einem Spectralon mit standardisierter Wellenlängeneigenschaften stattgefunden. Mittels „Continuum Removal“ ist eine konvex-einhüllende Funktion an die Spektren angepasst worden, um die Absorptionsbanden besser erkennbar zu machen (Jarmer et al., 2005). Nach Viscarra Rossel und Behrens (2010) absorbieren aromatische Kohlenstoffverbindungen, welche als stabil betrachtet werden, im Wellenlängenbereich von ca. 1650 nm, weshalb für die weitere Analyse nur noch der Wellenlängenbereich zwischen 1632 und 1669 nm verwendet worden ist. Die Spektraldaten sind einer agglomerativen, hierarchischen Clusteranalyse unterzogen worden (Ward, 1963), um die Proben nach der Durchsicht des Dendrogramms drei Gruppen zuweisen zu können.

Cluster 1 beinhaltet Bodenmaterial (N=32), welches eine mittlere Aromatizität aufweist. Im Cluster 1 (Reflexion zwischen 0.965 und 0.984) sind alle Alterklassen gleichanteilig vertreten, wobei nur drei Proben aus dem Horizont 0-5 cm stammen (Barren Land, Grass50 und Birch20). Cluster 2 repräsentiert ebenfalls Bodenproben aller Alters- und Tiefenkassen (N=38), wobei die meisten Proben von Birken- und Grasflächen und tieferen Beprobungshorizonten stammen. Cluster 2 weist die geringste Aromatizität (Reflexion zwischen 0.982 und 0.999) auf. Cluster 3 (N=14) beinhaltet Proben aller Tiefenklassen von Barren Land sowie Material aus den Beprobungsschichten 10-20 cm und 20-30 cm von Birch15, Birch20 und Birch50. Die Aromatizität ist bei Cluster 3 am höchsten (Reflexion zwischen 0.942 und 0.964).

Die Beziehung zwischen der Gesamtkohlenstoffkonzentration (C_{tot}) und der Kohlenstoffkonzentration in der S+C-Fraktion unter Berücksichtigung der Clusterbildung der Daten zeigt ein Muster auf, welches mit der Landbedeckungsänderung und dem Eintrag von Kohlenstoff in den Boden in Verbindung gebracht werden kann (Abbildung 28). Die Zunahme des SOC in der S+C-Fraktion korreliert grössten Teils mit der C_{tot}-Konzentration. Doch ab einer C_{tot}-Konzentration von 40 mg C g^{-1} Boden steigt die Zunahme der C-Konzentration in der S+C-Fraktion weniger stark. Die Daten in diesem Bereich mit geringerer Zunahme in der S+C Fraktion können eindeutig Cluster 2 zugeordnet werden. Diese Proben weisen die geringste Aromatizität auf und stammen von mit Vegetation bewachsenen Standorten. Weiter weisen die Proben von Cluster 3 (höchste Aromatizität) generell die tiefsten C_{tot}- und S+C-Konzentrationen auf. Diese Werte korrelieren sehr stark positiv (r=0.86, p<0.001). Dasselbe Verhalten (r=0.90, p<0.001) zeigen die Proben, welche Cluster 1 (mittlere Aromatizität) zugeordnet worden sind. Jedoch weisen sie etwas höhere C_{tot}- und S+C-Konzentration auf, was auf den höheren Kohlenstoffgehalt in den höheren Beprobungsschichten bei Cluster 1 als bei Cluster 2 zurückzuführen ist. Die Analyse kommt somit zum Schluss, dass Proben, welche aufgrund der höchsten Aromatizität eine hohe Stabilität des Kohlenstoffs aufweisen, die tiefsten Kohlenstoffgehalte aufweisen und nicht aufgeforsteten Flächen oder tieferen Beprobungsschichten zugewiesen werden. Weiter kann die Zunahme des Kohlenstoffgehaltes in der S+C-Fraktion infolge der Aufforstung mit diesem Denkansatz nicht mit einer Zunahme des aromatischen, stabilen Kohlenstoffs in Verbindung gebracht werden kann. Die Analyse deutet darauf hin, dass die Zunahme der S+C-Kohlenstoffkonzentration nicht mit einer korrelierenden Zunahme des stabilen Kohlenstoffs innerhalb der Fraktion in Verbindung gebracht werden kann, was die Eingangs in Kapitel 6.2 aufgestellte Vermutung, dass die Stabilität bei einer Zunahme der S+C-C Konzentration abnehmen kann, unterstützt.

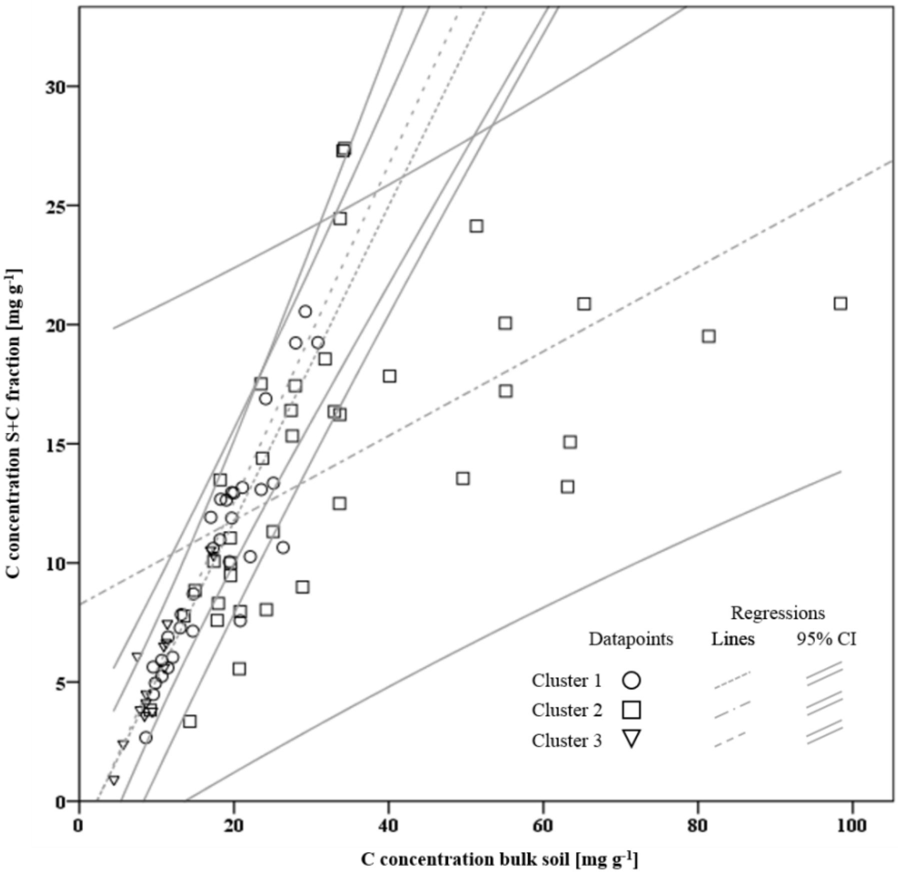


Abbildung 28: Beziehung zwischen Gesamtbodenkohlenstoffkonzentration [mg g^{-1}] und der Kohlenstoffkonzentration in der S+C-Fraktion [mg g^{-1}]. Die Daten sind mit Hilfe der drei Cluster, welche auf Basis der Spektraleigenschaften der 84 Proben ermittelt worden sind, klassifiziert worden.

6.3 Weiterverwendung des Probematerials

Die vorliegende Dissertation hat nebst Feinbodenmaterial, auch fraktioniertes Probematerial im Rahmen der drei Fallstudien erzeugt. Gegenwärtig beträgt der Probeumfang von den Fallstudien „Landaufgabe“, „Aufforstung“ und „Klimaerwärmung“ 180, 84 und 69 Proben (zu > 95% aus Mischproben (N=5) bestehend), welche während der physikalischen Fraktionierung in die drei Feststofffraktionen „POM“, „HF“ und „S+C“ separiert worden sind. Es bietet sich nun an, auf Basis dieser wertvollen Datengrundlage und neuer Forschungsfragen, die qualitativen Veränderungen des SOC infolge des Landbedeckungswandels mittels weiterer Analysemethoden zu untersuchen. Dabei stellen die genauere Untersuchung der Zusammensetzung des Kohlenstoffs in der „S+C“-Fraktion sowie die Interaktion des Kohlenstoffs mit der mineralischen der Schluff- und Tonphase erste Forschungsgegenstände dar. Generell bieten sich folgende Methoden oder Kombinationen davon an:

- Spektroskopische Untersuchung: Identifizierung von Kohlenstoffkomponenten und semi-quantitative Aussage auf Basis der Absorptionseigenschaften in den Wellenlängenbereichen des nahen Infrarot-, sichtbaren oder ultravioletten Lichts. Die Überlagerung der Absorptionsbanden des organischen Kohlenstoffmaterials durch Banden anderer Bodenbestandteile wie Wasser oder Mineralien muss bedacht werden (Viscarra Rossel et al., 2006; Zimmermann et al., 2007b; Stenberg et al., 2010).
- Analyse von Kohlenstoff- und Stickstoffisotopen: Aussage über die Dynamik (Stabilität, Verweilzeit und Zersetzungsrate) des Kohlenstoffs im Boden oder in einzelnen Fraktionen mittels Analyseverfahren auf Basis der Isoptope ^{12}C, ^{13}C, ^{14}N, ^{15}N (Balesdent et al., 1987; Liao et al., 2006; Conen et al., 2008; De Clercq et al., 2015).
- Kernspinresonanz (NMR): Bestimmung des Einflusses des Landbedeckungswandels auf die organische Bodensubstanz und deren Zersetzung sowie Stabilisierung (Baldock et al., 1992; Golchin et al., 1994; Baldock et al., 1997; Leifeld and Kögel-Knabner, 2005).
- Kalorimetrische Analyse: Quantifizierung von unterschiedlich stabilem Kohlenstoff hinsichtlich der exothermischen Regionen einzelner Kohlenstoffverbindungen (Dell’Abate et al., 2002; Plante et al., 2009)

KAPITEL 7

Synthese und Fazit

Durch die Temperaturerwärmung wird die Ausbreitung der Buschvegetation prognostiziert. Als eine weitere Folge davon erhofft man sich eine Diversifizierung der Landwirtschaft. Diese findet im Südwesten des Landes statt und konzentriert sich momentan auf die Schafzucht mit Stallhaltung während den Wintermonaten und extensiver Weidehaltung während des kurzen Sommers. In dieser Zeit wird auf den Flächen um die Höfe Winterfutter produziert. Wie lange sich unter diesen klimatischen Bedingungen aber die Lachse und Eisberge weiterhin in dieser Bucht im Erikfjord aufhalten werden, ist ungewiss. Aufgenommen von M. Hunziker am 16. Juli 2014.

7.1 Synthese

Waldökosysteme stellen ein grosses Senkenpotential hinsichtlich der Reduktion des atmosphärischen CO_2 dar, weil sie flächenmässig der grösste Landbedeckungstyp darstellen, 75 % der Bruttoprimärproduktion der Biomasse ausmachen und deshalb die höchste Kohlenstoffspeicherungsrate (total 2.4 GtC a^{-1}) von allen Landbedeckungstypen aufweisen (Beer et al., 2010; FAO, 2010; Pan et al., 2011). Folglich ist der grösste Teil des terrestrischen Kohlenstoffs in den Waldsystemen (860 GtC) gespeichert (Ciais et al., 2013; Ussiri and Lal, 2017a). Der Boden bildet im aktiven Kohlenstoffkreislauf ein grosses Kohlenstoffreservoir (total 1500 bis 2400 GtC), indem der von der Vegetation stammende Kohlenstoff über längere Zeit darin gespeichert werden kann (Lorenz, 2013). Aufgrund der Grössenverhältnisse der beiden Reservoire „Boden" und „Atmosphäre" (840 GtC) hat eine geringe Zunahme des Kohlenstoffvorrates im Boden eine bedeutende Verringerung der Menge an atmosphärischem CO_2 zur Folge (Lal, 2004). Deshalb gilt die Afforstung von vormals intensiv genutzten Gebieten als erfolgreiche Massnahme, um die Auswirkungen der anthropogenen Treibhausemissionen zu lindern (IPCC, 2014b; Ussiri and Lal, 2017d).

Aktiv durch den Menschen induzierten Landbedeckungsänderungen hin zu Waldvegetation sind in den letzten beiden Jahrzehnten hinsichtlich den quantitativen und qualitativen Bodenkohlenstoffveränderungen untersucht und evaluiert worden (Guo and Gifford, 2002; Vesterdal et al., 2002; Cerli et al., 2006; Laganière et al., 2010; Poeplau et al., 2011; Vesterdal et al., 2011, 2013; Poeplau and Don, 2013; Hiltbrunner et al., 2013; Bárcena et al., 2014a, 2014b; Guidi et al., 2014b). Die Studien haben hervorgebracht, dass die vormalige Landbedeckung (z.B. Ackerland, Grasland, Heideland oder Boden ohne Vegetation) sowie der Waldtyp (Nadel-, Laub- oder Mischwald) mitbestimmend über den Verlauf der Bodenkohlenstoffvorratsveränderung (C-Senke oder C-Quelle während den untersuchten Zeitspannen) sind. Eine Zunahme des Kohlenstoffvorrates ist in den wenigsten Fällen während den ersten Jahren nach dem Landnutzungswechsel festgestellt worden. Die Analyse der qualitativen Kohlenstoffveränderung hat eine Zunahme der Kohlenstoffkonzentration in der POM-Fraktion und eine Abnahme der Konzentration in den Fraktionen, in denen die organische Substanz mit der mineralischen Phase verbunden ist, gezeigt. Folglich führt die Aufforstung zu einem Anstieg der Labilität der organischen Substanz im Boden (Poeplau and Don, 2013; Guidi et al., 2014a). In all den Studien ist der Fokus auf in forstwirtschaftlicher Sicht produktiven Waldgesellschaften gelegen, welche unter anderem auch die Definitionskriterien der FAO von Wald erfüllen (FAO, 2006a).

Studien zeigen aber, dass die Flächen der Buschvegetation, welche die Definitionskriterien von Wald nicht erfüllen, beispielsweise in den Alpen, in Grönland, in Alaska oder Island ebenfalls zunehmen (Anthelme et al., 2002; Tape et al., 2006; Drees and Daniëls, 2009; Myers-Smith et al., 2011; Huber and Frehner, 2013; Caviezel et al., revised). Diese Buschzunahme findet in marginalen Landschaftsräumen statt, welche einen sehr geringen oder keinen Nutzungsdruck aufgrund der Entlegenheit im zirkumpolaren Raum oder veränderten Nutzungsansprüche, welche zu einer Extensivierung von topographisch schwer zugänglichen Gebieten verursacht hat (Tasser and Tappeiner, 2002; Streifeneder, 2009; Tasser et al., 2011).

Die vorliegende Dissertation hat sich deshalb dem Kohlenstoffverhalten im mineralischen Boden (0-30 cm) in diesen Grenzräumen, in denen die Verbuschung der Landschaft festgestellt wird, gewidmet und dabei die quantitativen und qualitativen Eigenschaften des Bodenkohlenstoffes untersucht. Zur qualitativen Beurteilung ist eine Kombination der Grössen- und Dichtefraktionierung des Bodens angewendet worden. Die Beschreibung der Veränderung der Bodenkohlenstoffqualität infolge des Landbedeckungswandels durch Buschvegetation in alpinen und hochatlantischen, subarktischen Grenzräumen ist bis anhin nicht untersucht worden, was den Wert der Resultate der vorliegenden Dissertation unterstreicht. Als Grenzräume sind ausgewählt worden: Die subalpine Vegetationsstufe zwischen dem Hochwald und dem alpinen Rasen sowie der „mountain birch belt“ im nordatlantischen Raum, als Teil der zirkumpolaren Buschtundra, welche als Grenzökoton zwischen den beiden Biomen Boreal und Tundra gilt (Payette et al., 2001; Wielgolaski, 2001). Folgende drei Landschaftssysteme gekoppelt mit drei verschiedenen Ursachen der Landbedeckungsänderung sind dabei betrachtet worden:

i) Ausbreitung der Grünerle (*Alnus viridis* (Chaix) DC.) infolge der Extensivierung und Landaufgabe von subalpinen Alpweiden in der Zentralschweiz (Kapitel 2 und 3),
ii) Das Aufkommen der Moorbirke (*B. pubescens* Ehrh.) auf stark degradierten Böden in Island infolge der Aufforstung (Kapitel 4),
iii) Auswirkung der Klimaerwärmung auf die Ausbreitung der Moorbirke (*B. pubescens* Ehrh.) als Bestandteil der Waldtundra im Boreal-Tundra Grenzökoton in Südwestgrönland (Kapitel 5).

Im Unteralptal hat die mit *Alnus viridis* (Chaix) DC. bewachsene Fläche zwischen 1959 und 2007 um 87 ha (+ 63 %) zugenommen. Die räumliche und zeitliche Analyse der Verbuschungsdynamik zeigt weiter auf, dass dabei auch subalpine Alpweiden an strahlungsintensiveren Südwest- bis Südosthängen oder mit geringerer Hangneigung als 60 % verbuscht werden. Die Studie (Kapitel 2) kommt daher zum Schluss, dass die ökologische Nische von *Alnus viridis* (Chaix) DC. grösser ist, als bisher angenommen und die Landaufgabe nebst dem Relief ein entscheidender Faktor bei der Ausbreitung von *A. viridis* ist. Aus diesem Grund können potentielle Verbuschungsflächen mit den Angaben aus der Literatur nur ungenügend vorhergesagt werden und der Lebensraumtyps „*Alnenion viridis*“ (Delarze et al., 2015) hat flächenmässig einen grösseren Einfluss auf das subalpine Bodensystem. Die Veränderung der bodenphysikalischen Eigenschaften und der Einfluss auf die Bodenstabilität infolge der Verbuschung sind bereits aufgezeigt worden (Caviezel et al., 2014). Hunziker et al.

(2017) (Kapitel 3) zeigt auf, dass das Einwachsen von *A.viridis* auf subalpinen Alpweiden ebenfalls den Bodenkohlenstoffvorrat quantitativ und qualitativ verändert. Während den ersten 40 Jahren der Verbuschung durch *A. viridis* nimmt der Gesamtkohlenstoffvorrat (0-30 cm) der Alpweiden von 100 t C ha^{-1} auf 81 t C ha^{-1} ab, weshalb der Boden in diesem Zeitraum als C Quelle (0.48 t C ha^{-1} a^{-1}) agiert (Abbildung 29; A). Bis zu diesem Zeitpunkt der Verbuschung ist die Abnahme des mineralischen Kohlenstoffvorrats mit den Ergebnissen anderer Studien vergleichbar (Thuille and Schulze, 2006; Alberti et al., 2008; Hiltbrunner et al., 2013; Guidi et al., 2014a) und typisch für die Änderung von Grasland zu Wald- resp. Buschsystemen (Poeplau et al., 2011; Bárcena et al., 2014). Jedoch beträgt der Kohlenstoffvorrat (0-30 cm) nach 90 jährigem Grünerlenwachstum und der Bildung des Lebensraumtyps „*Alnenion viridis*" 174 t C ha^{-1}, was einer signifikanten Erhöhung des Kohlenstoffreservoirs um 74 % im Vergleich zu jenem der Alpweide (v.a. *Poion alpinae* Lebensraumtyp (Delarze et al., 2015)) entspricht (Abbildung 29; A). Der Boden stellt somit zwischen 40 und 90 Jahren nach dem Landbedeckungswandel eine C-Senke dar (1.86 t C ha^{-1} a^{-1}). Über den Zeitraum von 90 Jahren betrachtet, beträgt die jährliche Kohlenstoffzunahme 0.86 t C ha^{-1}. Die Zunahme des SOC Vorrates von 74 % innerhalb von 90 Jahren steht im Widerspruch zu der modellierten „carbon response function" für den Wechsel von Grasland zu Wald gemäss Poeplau et al. (2011). Denn die Prognose sagt für 100 Jahre nach dem Wechsel einen um 7% ± 23% tieferen mineralischen SOC Vorrat voraus. Die Funktion kann somit für die Untersuchung des Kohlenstoffvorrates infolge der Verbuschung durch *A. viridis* nicht zur Anwendung gebracht werden. Weiter dauert es laut der besagten Funktion mehr als 150 Jahre, bis der Kohlenstoffvorrat des mineralischen Bodens das Niveau des Ausgangszustandes erreicht hat. Dies trifft im Fall der Verbuschung durch *A. viridis* im subalpinen Raum nicht zu. Es dauert deutlich weniger lang als 90 Jahre (Abbildung 29; A).

Die Hochrechnung der Bodenkohlenstoffmenge auf die abgeschätzte, durch Grünerlen zugewachsene Alpweidenfläche, welche aus den Nationalen Forst Inventaren (NFI) 83/85, 93/95 und 04/06 berechnet wurde, zeigt deutliche Unterschiede zwischen der Andwendung des im NIR benutzten Standardkohlenstoffvorrates von 69 t C ha^{-1} (FOEN, 2015) und den altersabhängigen Vorräten der Fallstudie „Landaufgabe" (84 und 91 t C ha^{-1}). Die Menge an Kohlenstoff wird mit dem NIR-Wert bei 25-jährigen und 15-jährigen Grünerlenflächen um $4x10^3$ t C (18 %) und $97x10^3$ t C (24 %) unterschätzt (Tabelle 17).

Tabelle 17: Hochrechnungen der Menge an Bodenkohlenstoff [1000 t C] (0-30 cm) mit Hilfe des Vorrats pro Hektare, der im NIR (69 t C ha^{-1}) (FOEN, 2015) verwendet wird, und der Vorräte für 15 (84 t C ha^{-1}) und 25 (91 t C ha^{-1}) jährige Grünerlenstandorte aus Kapitel 3, welche kompatibel mit den NFI Perioden 83/85 – 93/95 und 93/95 – 04/06 sind. Die Fläche des Gebüschwaldes betrifft nur die Zuwächse zwischen den NFI Zeitständen und die Grünerlenfläche ist mit dem Faktor 0.7 ermittelt worden (Brändli, 2010). Es fand keine Klassifzierung nach der Höhe statt.

NFI Periode	Fläche in den „Alpen“ und „Südalpen“		Kohlenstoffvorrat		Differenz
	Gebüschwald	Grünerlen	NIR	Kapitel 3	
	[1000 ha]	[1000 ha]	[1000 t C]	[1000 t C]	[1000 t C]
83/85 – 93/95	0.40	0.28	19.32	23.52	4.20
93/95 – 04/06	6.30	4.41	304.29	401.31	97.02

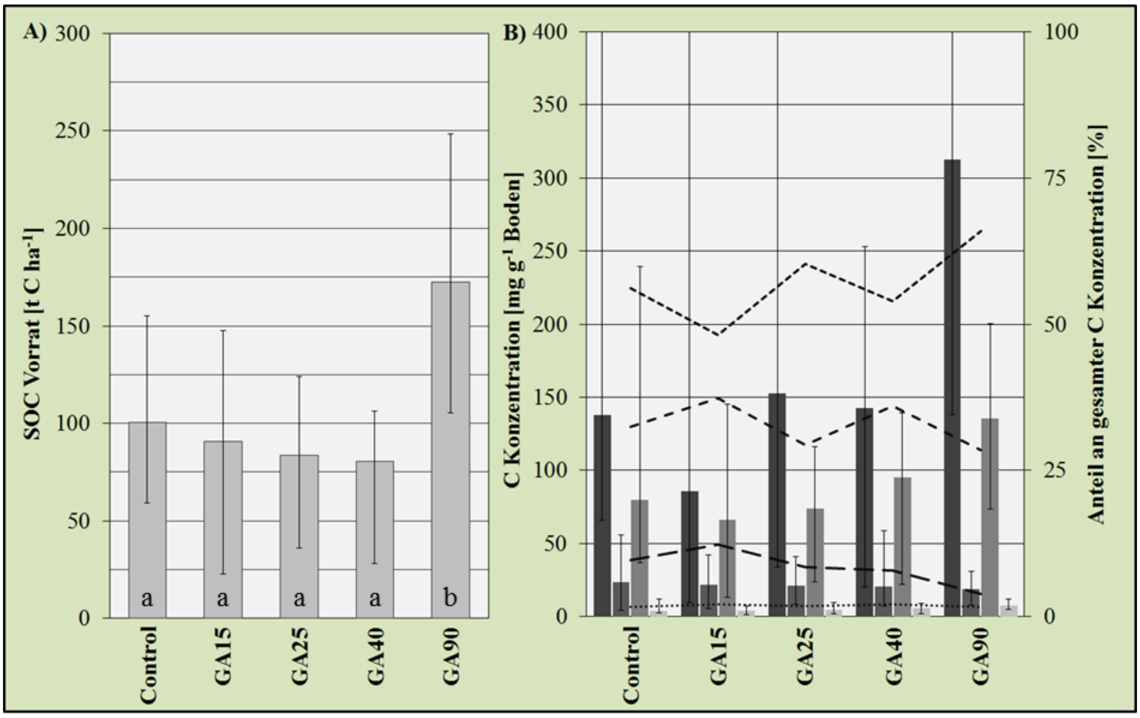


Abbildung 29: Veränderungen des Kohlenstoffvorrats (0-30 cm) [t C ha^{-1}] (a) sowie der Kohlenstoffkonzentration (0-30 cm) [mg g^{-1} Boden] in den untersuchten Fraktionen POM (schwarz), HF (dunkelgrau), S+C (grau), DOC (hellgrau) und den relativen Veränderungen [%] der Fraktionen POM (fein gestrichelt), HF (lang gestrichelt), S+C (gestrichelt) und DOC (gepunktet) (b) im Laufe der Verbuschung durch *A. viridis* infolge der Landaufgabe. Die X-Achse zeigt den zeitlichen Verlauf der SOC Veränderungen unter Alpweiden (Control) und unterschiedlich alten Grünerlenbeständen (15, 25, 40 und 90 Jahre) auf. Die Fehlerbalken zeigen die Bereiche zwischen Minimum- resp. Maximum- und Medianwerten (N=9) an. Die Maximal-Werte in der POM-Fraktion sind bei „Control“, „GA15“, „GA25“ und“GA90“ 772, 466, 470 und 1161 mg g^{-1} Boden. Signifikante Unterschiede zwischen den Alterklassen pro SOC-Fraktion sind in Abbildung 13 aufgeführt.

Das Verhalten des Kohlenstoffs in den untersuchten Fraktionen weist zudem auf eine Erhöhung der Labilität des Bodenkohlenstoffes hin (Abbildung 29; B). Nebst der signifikanten Zunahme der SOC Konzentrationen der POM- und DOC-Fraktionen (Abbildung 13), verdoppelt sich die Gewichtsmasse in der POM Fraktion zwischen Alpweiden und 90 jährigen Grünerlenstandorten, was zu einem Kohlenstoffvorrat in der POM Fraktion von 115 t C ha^{-1} führt und bereits mehr als der Mediankohlenstoffvorrat (100 t C ha^{-1}) für Böden in den Alpen ist (Sjögersten et al., 2011). Die Vulnerabilität des SOC erhöht sich mit der Verbuschung zusätzlich, weil die Konzentration in der „Sand+Aggregate“ (HF) Fraktion abnimmt und die Zunahme der C Konzentration in der „Schluff+Ton“ (S+C) Fraktion zwar zunimmt, aber wie in Kapitel 6.2.1 diskutiert und aufgezeigt worden ist (Abbildung 27), der Kohlenstoff in der Fraktion auch als partikuläre organische Substanz (< 63 µm) vorliegen kann. Der Vergleich der prozentualen Anteile der C-Konzentrationen der einzelnen Fraktionen im Verhältnis zur Gesamtkohlenstoffkonzentration im Boden deutet durch den Anstieg der POM- und DOC-Anteile und den Abnahmen der HF- und S+C-Anteile auf einen Anstieg der SOC Vulnerabilität durch die Etablierung des *Alnenion viridis* auf subalpinen Alpweiden innerhalb von 90 Jahren hin (Abbildung 29; B).

Die Aufforstung in Island hat nebst der Steigerung anderer Ökosystemleistungen auch die Zunahme des Bodenkohlenstoffvorrats zum Ziel, weil die stark degradierten Böden ein Kohlenstoffspeicherungspotential von jährlich 0.6 bis 1.0 t C ha^{-1} aufweisen (Arnalds et al., 2000; Gudmundsson et al., 2004). In der Studie (Kapitel 4) ist das Bodenkohlenstoffverhalten (0-30 cm) von mit *B. pubescens* Ehrh. aufgeforsteten Standorten mit jenem von stark degradierten Böden, welche den Ausgangszustand darstellen, und nicht degradierten Böden von natürlich gewachsenen Birkenbuschland (angestrebter Endzustand) verglichen worden. Der untersuchte degradierte Boden weist einen um 20 t C ha^{-1} tieferen Vorrat auf als jener mit ungestörter und natürlich gewachsener Birkenbuschvegetation (59 t C ha^{-1}) (Abbildung 30; A), was das Potential der C-Speicherung von vegetationslosen Böden bestätigt (Bárcena et al., 2014). Die Etablierung von Buschwaldvegetation infolge der Aufforstung zeigt eine kontinuierliche Zunahme des Bodenkohlenstoffvorrats von 31 t C ha^{-1} auf 46 t C ha^{-1} zwischen 15 und 50 jährigen Buschbeständen auf. Der angestrebte SOC Vorrat von 59 t C ha^{-1} ist somit nach 50 Jahren Birkenwachstum und der Speicherung des von der Biomasse stammenden Kohlenstoffs im Boden noch nicht erreicht (Abbildung 30, A). Weiter dient der Boden während mindestens dieser Zeitdauer als C-Senke. Die Degradierung der Landschaft und die vorherrschende Bodenerosion in historischer Zeit haben dazu geführt, dass der Ausgangszustand der zu rekultivierenden Böden sehr heterogen ist. Daher stellt der Chronosequenz-Ansatz unter solchen Voraussetzungen nicht das geeignetste Beprobungsschema dar. Deshalb ist es zweifelhaft, den Kohlenstoffvorrat der degradierten Böden, welcher den Zustand vor der Rekultivierung und Aufforstung darstellt, mit den Vorräten der aufgeforsteten Flächen zu vergleichen. Denn die errechneten SOC Speicherungsraten mit den SOC Vorräten der degradierten Böden als Basis liegen tiefer als die Annahmen aus der Literatur (Bjarnadottir, 2009; Hellsing et al., 2016) oder sind gar negativ, obwohl ein Zuwachstrend der SOC Vorräte mit Zunahme des Alters der aufgeforsteten Flächen messbar ist (Abbildung 16, Abbildung 30; A). Die Studie zeigt auf, dass in tieferen Bodenschichten die

Bodenkohlenstoffvorräte höher sein können als nahe der Geländeoberfläche, was zu den negativen SOC Speicherungsraten führt.

Die Entstehung von Birkenbuschflächen auf vormals degradierten, vegetationslosen Böden führt dazu, dass die SOC Konzentration in der POM-Fraktion am stärksten zunimmt und bei 50 jährigen Birkenbeständen im Vergleich zu jener von natürlich gewachsenen Birkenwäldern noch höher liegt (Abbildung 30; B), was auf die unterschiedliche Produktivität der Birkenvegetation in den verschiedenen Entwicklungsstadien zurückzuführen ist (Smith et al., 1997). Die C Konzentrationen in den mineralischen SOC Fraktionen (HF und S+C) nimmt während des Aufkommens von Birkenvegetation zu, was auf eine Stabilisierung des Bodenkohlenstoffes schliessen lässt. Die Art der Stabilisierung des Kohlenstoffs in der S+C-Fraktion benötigt weitere Untersuchungen, insbesondere im Zusammenhang mit den Stabilisierungsmechanismen der vulkanischen Tonmineralien. Der Vergleich des SOC zwischen den einzelnen Fraktionen zeigt auf, dass trotz absolutem Anstieg der Konzentrationen in der HF- und S+C-Fraktion eine Stagnation in der HF-Fraktion resp. Abnahme in der S+C-Fraktion des relativen Anteils des SOC in den mineralischen Fraktionen im Verhältnis zum Anstieg des relativen SOC Anteils in der POM-Fraktion während der Entstehung von Birkenbuschwald stattfinden (Abbildung 30; B). Die Aufforstung auf degradierten Böden mit *B. pubescens* Ehrh. führt somit in den ersten 50 Jahren zu labileren Bodenkohlenstoffbedingungen. Die Resultate zeigen darüber hinaus, dass die Entwicklung der Standorte ab 50 Jahren Birkenwachstum wieder zu stabileren Bodenkohlenstoffbedingungen führen kann, weil der angestrebte Gleichgewichtszustand innerhalb von 50 Jahren noch nicht erreicht worden ist (Abbildung 30; B). Mit Hilfe der Fraktionierung der Isländischen Böden ist ersichtlich geworden, dass in den degradierten Böden über zwei Drittel des Kohlenstoffs in der S+C-Fraktion vorliegen. Somit ist der Kohlenstoff in der S+C Fraktion für die hohen Bodenkohlenstoffvorräte in degradierten Böden verantwortlich, was bei der Inventarisierung des Bodenkohlenstoffes unter Berücksichtigung des Landbedeckungswandels in Zukunft bedacht werden muss.

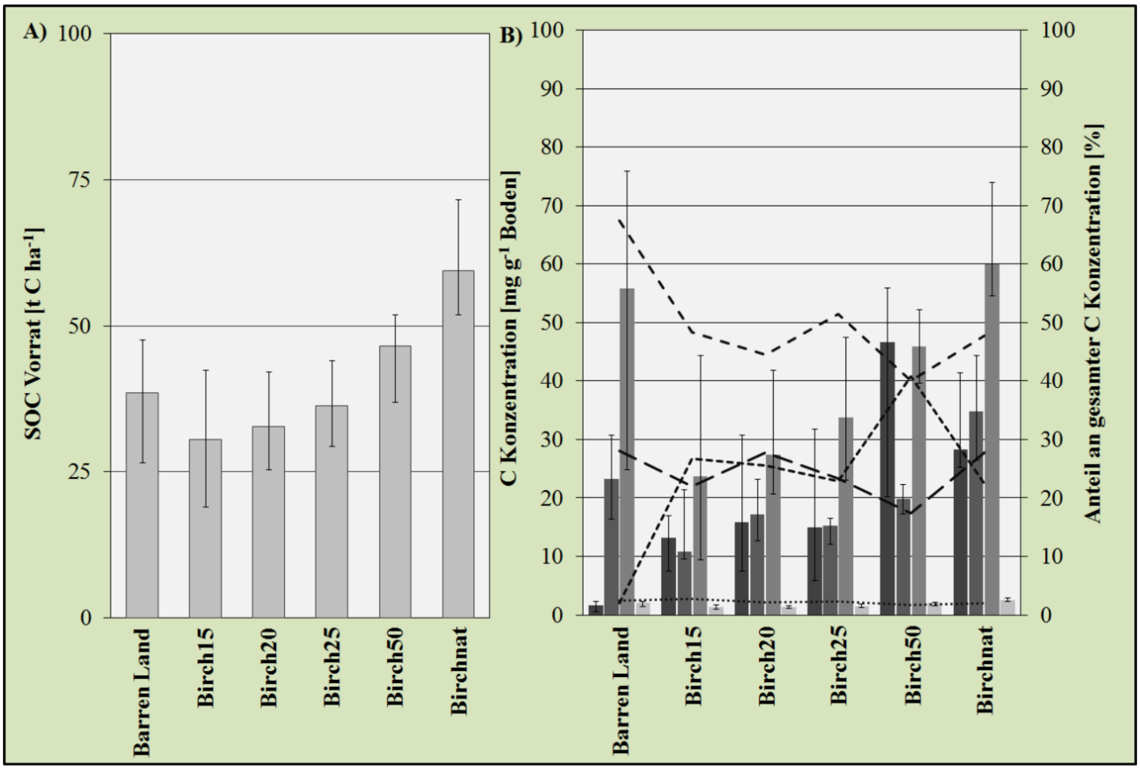


Abbildung 30: Veränderungen des Kohlenstoffvorrats (0-30 cm) [t C ha^{-1}] (a) sowie der Kohlenstoffkonzentration (0-30 cm) [mg g^{-1} Boden] in den untersuchten Fraktionen POM (schwarz), HF (dunkelgrau), S+C (grau), DOC (hellgrau) und den relativen Veränderungen [%] der Fraktionen POM (fein gestrichelt), HF (lang gestrichelt), S+C (gestrichelt) und DOC (gepunktet) (b) im Laufe der Etablierung von *B. pubescens* infolge der Aufforstung. Die X-Achse zeigt den zeitlichen Verlauf der SOC Veränderungen in Böden in stark degradiertem, vegetationslosem Zustand (Barren Land), unter unterschiedlich alten aufgeforsteten Birkenbeständen (15, 20, 25 und 50 Jahre) sowie unter natürlich gewachsener, alter Birkenvegetation. Die Fehlerbalken zeigen die Bereiche zwischen Minimum- resp. Maximum- und Medianwerten (N=3) an.

Die Klimaerwärmung von 2.5 °C in Südwestgrönland während den letzten 110 Jahren (Hanna et al., 2012) sowie die prognostizierte zusätzliche Erwärmung um 3.3 °C bis 2100 (Masson-Delmotte et al., 2012) sind vergleichbar mit den Messungen und Prognosen für den zirkumpolaren Raum. Durch die Temperaturerhöhung wird eine Verschiebung der Vegetationsgrenzen im Übergangsbereich zwischen den Biomen Tundra und Boreal vorhergesagt (Myers-Smith et al., 2011). In Südwestgrönland wird dies mit der Zunahme der Buschvegetation in der Zone, welche von *B. pubescens* Ehrh. dominiert wird, in Verbindung gebracht (Fredskild and Odum, 1990; Normand et al., 2013). Die Studie (Kapitel 5) hat die Bodenkohlenstoffeigenschaften von Buschvegetation und buschloser Tundravegetation auf Einzugsgebietsebene verglichen und dabei abgeschätzt, welche Auswirkungen eine Zunahme der Buschvegetation infolge der Klimaerwärmung auf den Bodenkohlenstoff hat. Die Resultate zeigen, dass die Kohlenstoffvorräte (0-30 cm) von Birkenbuschvegetation und buschloser Vegetation zwischen 54 und 148 t C ha^{-1} (Medianwerte) variieren (Abbildung 31; A). Nebst der Art der Vegetationsbedeckung, welche etwas höhere SOC Vorräte bei der Birkenvegetation zur Folge hat, sind die Unterschiede im Kohlenstoffvorrat mehr durch die Lage der untersuchten

Vegetationsstandorte in der Landschaft zu erklären. Diese beeinflusst den Biomassevorrat in der Vegetation und somit das Angebot an Kohlenstoff für den SOC Vorrat. Der Bodenkohlenstoff wird vorwiegend in der POM- und S+C-Fraktionen gespeichert, wobei die POM-Fraktion bei Birkenstandorten und die S+C-Fraktion bei buschlosen Vegetationsstandorten dominieren (absolute Werte und relative Anteile) (Abbildung 31; B). Die POM-Fraktion kann aber auch bei buschloser Vegetation (Shrub-free (W) in Abbildung 31; B) einen ähnlich hohen Anteil an Kohlenstoff wie in der S+C-Fraktion aufweisen, weil die Standorteigenschaften eine Zersetzung der organischen Substanz und Einarbeitung in die mineralische Phase hindern können (Abbildung 20, Abbildung 31; B). Der relative Anteil an C in der HF-Fraktion liegt bei den untersuchten Standorten jeweils unter 20 Prozent. Eine Ausbreitung der Birkenvegetation kann an Standorten, welche für das Wachstum günstig sind, zu einer Erhöhung des Bodenkohlenstoffvorrates führen, was aber mit einer Zunahme der Labilität des SOC verbunden ist.

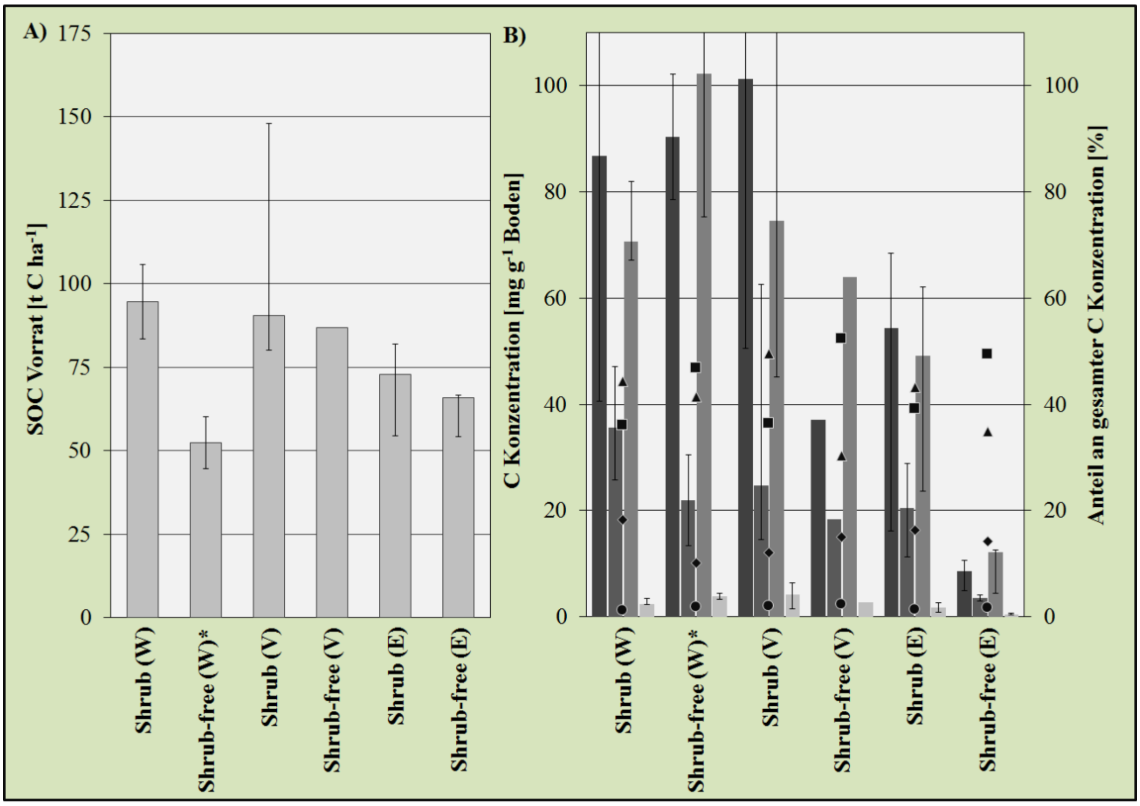


Abbildung 31: Verhalten des Kohlenstoffvorrats (0-30 cm) [t C ha^{-1}] (a) sowie der Kohlenstoffkonzentration (0-30 cm) [mg g^{-1} Boden] in den untersuchten Fraktionen POM (schwarz), HF (dunkelgrau), S+C (grau), DOC (hellgrau) und den relativen Veränderungen [%] der Fraktionen POM (Dreieck), HF (Diamant), S+C (Quadrat) und DOC (Kreis) (b). Die X-Achse zeigt die Unterteilung nach den drei untersuchten Landschaftseinheiten „West“ (W), „Valley“ (V) und „East“ (E) sowie die beprobten Buschbestände mit *B. pubescens* (Shrub) und Tundravegetation ohne Buschvorkommen (Shrub-free). Die Fehlerbalken zeigen die Bereiche zwischen Minimum- resp. Maximum- und Medianwerten an. Die Maximal-Werte sind bei „Shrub (W)“: 122 (POM), „Shrub-free (W)*“: 129 (S+C) und „Shrub (V)“ 231 (POM) und 112 (S+C) mg g^{-1} Boden. Die Anzahl Messwerte pro Klasse variiert zwischen N=1 und N=7. Das Beprobungsinterval 0-10 cm bei „Shrub-free (W)“ ist mit einem Stern (*) gekennzeichnet.

Die Studie kommt darüber hinaus zum Schluss, dass das Wirkungsgefüge verschiedener Geoökofaktoren für das Vorhandensein von Buschvegetation und die Quantität und Qualität des Bodenkohlenstoffes im Untersuchungsgebiet verantwortlich ist. Dieses Wirkungsgefüge ist zusammen mit der regionalen Klimaerwärmung auch für die Ausbreitung der Buschvegetation mitverantwortlich. Deshalb sollten Klima-Vegetations–Modelle, die mit klimatischen Variablen als einzige Steuergrössen die künftige Ausbreitung der Buschvegetation prognostizieren zu versuchen, um weitere Geoökofaktoren als Modellvariablen ergänzt werden. Auf Einzugsgebietsebene sind dabei Relief und lithologische Zusammensetzung des anstehenden Gesteins als Hauptfaktoren ausfindig gemacht worden.

7.2 Fazit

Hinsichtlich der Auswirkung der Landbedeckungsänderung durch das Aufkommen von Buschvegetation in alpinen und subarktischen Grenzräumen können nachfolgenden Aussagen bezüglich der Veränderung der Quantität und Qualität des Bodenkohlenstoffes gemacht werden.

Der Boden stellt durch das Aufkommen von Grünerlen auf aufgelassenen subalpinen Alpweiden in der Zentralschweiz in den ersten 40 Jahren des Buschwachstums eine C-Quelle und zwischen 40 und 90 Jahren eine C-Senke dar. Die Kohlenstoffspeicherung wirkt sich in einem um 74 t C ha^{-1} höheren Bodenkohlenstoffvorrat im Vergleich zu jenem der Alpweiden aus. Über 90 Jahre betrachtet beträgt die jährliche Speicherungsrate 0.86 t C ha^{-1}. Im subarktischen Raum zeigt der Vergleich der Bodenkohlenstoffvorräte von stark degradierten Isländischen Böden aus Substrat vulkanischen Ursprungs und jenen von geschützten, natürlich gewachsenen Birkenbuschwäldern ein SOC-Speicherpotential von 20 t C ha^{-1} auf. Während 35 Jahren werden im Boden von aufgeforsteten Birkenbeständen 15 t C ha^{-1} gespeichert, was einer C-Senke mit einer jährlichen Speicherrate von 0.4 t C ha^{-1} entspricht. Jedoch zeigt die Untersuchung ebenfalls auf, dass die vormals aktive Bodenerosion zu unterschiedlichen Bodenkohlenstoffvorräten in den aufzuforstenden Böden geführt hat. Diese können, wie in der Studie gezeigt, höhere Bodenkohlenstoffvorräte als bereits verbuschte Flächen aufweisen, was die Abschätzung des Senkenpotentials erheblich erschwert. Für die Ausbreitung der Buschtundra im subarktischen Boreal-Tundra Grenzökoton Südwestgrönland, welche infolge der Klimaerwärmung vorhergesagt wird, ist kein klarer Trend hinsichtlich C-Senke oder C-Quelle des Bodens festgestellt worden. Die Bodenkohlenstoffvorräte von Birkenbuschvegetation und buschloser Vegetation variiert innerhalb des Untersuchungsgebiets zwischen 54 und 148 t C ha^{-1}. Dabei sind die grössten Vorräte bei Buschvegetation an für das Buschwachstum günstigen Standorten (tiefergelegene Schutzlagen) gemessen worden.

Unabhängig von den Prozessen, die zu einem Aufkommen der Buschvegetation in marginalen Grenzräumen führen, zeigen die Resultate der drei Fallstudien, dass der Landbedeckungswandel die Quantität und Qualität des Kohlenstoffs im mineralischen Boden verändert. Bestehende „carbon response functions“ für die Umwandlung in Waldsysteme können für den Landbedeckungswandel zu Buschvegetation in alpinen und subarktischen Räumen nicht angewendet werden, weil Produktivität der Buscharten und vermutlich der Geoökofaktor Temperatur das Bodenkohlenstoffverhalten entscheidend beeinflussen. Alle drei Fallstudien zeigen, dass eine Zunahme des Kohlenstoffvorrates im mineralischen Boden infolge der Verbuschung möglich ist. Die Studien haben aber auch eine Zunahme der Kohlenstoffkonzentration sowie eine Zunahme des relativen Anteils des labilen Kohlenstoffs in der POM- und DOC-Fraktion gemessen, welcher das Bodenkohlenstoffsystem kurzfristig (innerhalb von Jahren bis wenigen Jahrzehnten) (von Lützow et al., 2008) verlassen kann. Weiter zeigen die relativen Veränderungen der Konzentrationen in den SOC Fraktionen eine Stagnation oder Abnahme des SOC in der HF-Fraktion. Die Arbeit hat nicht abschliessend beantworten können, ob mit den verzeichneten Zunahmen der C-Konzentration in der „Schluff- und Ton“-Fraktion während der Verbuschung durch *A. viridis* und während der Aufforstung mit *B. pubescens* Ehrh. auch eine Stabilisierung dieses Kohlenstoffs in dieser Fraktion einhergeht.

Der Bodenkohlenstoff verhält sich bei Vegetationsveränderungen dynamisch, was sich in messbaren Unterschieden der Kohlenstoffvorräten oder der C-Konzentrationen in den einzelnen SOC Fraktionen ausgedrückt hat. Auf Basis der vorliegenden Arbeit ist davon auszugehen, dass sich dabei die Labilität des Bodenkohlenstoffes in den sich neu etablierenden Ökosystemen innerhalb der untersuchten Grenzökotonen, welche durch Buschvegetation dominiert werden, erhöht. Trotz gemessenen Zunahmen der Kohlenstoffvorräte ist der Bodenkohlenstoff daher weniger resistent gegenüber Störungen wie die bodenbiologische Aktivität oder Feuerereignisse, die durch Klimaveränderungen (Temperaturerhöhung oder Trockenperioden) begünstigt werden.

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Wissenschaftliche Beiträge

Publikationen

2011	Hunziker, M. A study on above- and belowground biomass and carbon stocks as well as sequestration of mountain birch (Betula pubescens Ehrh.) along a chronosequence in southern Iceland. Masterarbeit, Universität Basel.
2013	Würsch, M., Carle, N., Hunziker, M. Bodendegradation und entgegenwirkende Massnahmen in Island – Beiträge der Basler Forschung. Regio Basiliensis (54), S. 11-18.
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Vorträge und Poster an Konferenzen (Hauptautor)

2010	Hunziker, M., Sigurdsson, B.D., Halldorsson, G., Kuhn, N.J. Woody Biomass and Carbon Stocks of Natural vs. Restored Mountain Birch (Betula pubescens Ehrh.) Woodlands in South Iceland. Geophysical Research Abstracts, Vol. 12, EGU2010-14776-1, Poster.
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Betreute Abschlussarbeiten

2012	Carle, Nina. Soil Fractions and their Relation to Functional Pools. Bachelorarbeit.
2013	Carle, Nina. The fate of soil organic matter – a chronosequence study investigating stability under reforestation. Masterarbeit.
2013	Würsch, Mathias. Land reclamation in Iceland: accumulation and distribution of soil organic carbon and nitrogen in fields of Lupinus nootkatensis. Masterarbeit.
2013	Wyss, Emanuel. Auswirkung der Vegetationsveränderung im Unteralptal auf die dortige Hydrologie im Zeitraum zwischen 1979 und 2007. Bachelorarbeit.
2014	Diesslin, Alexandra. Eine holzanatomische und dendroökologische Untersuchung des Verbuschungspotentials in Südwestgrönland. Bachelorarbeit.
2014	Zoller, Oliver. The influence of climate change and topography characteristics on the shrub encroachment in south Greenland. Masterarbeit.