Chapter 6
Carbon Stocks and Carbon Stock Changes in German Forest Soils

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6.1 Introduction

Forests play an important role in the global carbon (C) cycle. Through photosynthesis plants convert atmospheric carbon dioxide (CO_2) into plant biomass. About half of the C stored in the plant biomass is subsequently released into the atmosphere by respiration in plants. Most of the remaining C enters the soil, e.g., as dead leaf or root biomass. The amount of soil C is determined by the net balance of the rate of organic C input and its decomposition. Soils are the largest reservoir of organic C in the active C cycle of terrestrial ecosystems.

Globally, they contain about 1.5–2.0×10^{15} tons of organic C to a depth of 1 m (Amundson 2001). Forest soils store one third of the global organic C. They are a larger reservoir for C than plants and atmosphere combined (Schlesinger 1991).

With regard to climate protection policy, the sequestration of atmospheric C in soils as stable organic matter is discussed as a potential contribution to mitigate atmospheric CO_2 concentrations (Janzen 2004). A proportion of the C entering the soil is mineralized by microorganisms resulting in the release of CO_2 and/or

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methane. Carbon can also be leached from soil as dissolved or particulate organic matter soil C not mineralized by soil microorganisms—ranging from young recently deposited plant material to very old organic constituents in the so-called stable organo-mineral associations—is stored in the soil as organic matter. The soil organic matter fulfills important soil functions as it serves as a nutrient exchanger and buffer for plants, improves the retention of water, and increases the buffering and exchange capacity of soils. Soil organic matter also provides energy for heterotrophic soil organisms as well as for the microflora.

An increase in C stocks is expected to be a result of environmental as well as human-induced factors (Tan et al. 2004; Liski et al. 2002; de Vries et al. 2009). The mineralization of organic matter is limited by temperature and water availability, and thus the amount and dynamic of soil C pools are substantially affected by climate (Davidson and Janssens 2006; Lorenz and Lal 2010). Currently, it is not yet fully resolved why one part of soil C is decomposed rapidly, while other parts are characterized by turnover times ranging from decades to millennia (Baisden et al. 2002). One possible reason is that the accumulation of soil C is controlled by the restricted availability of decomposing organisms and their enzymes (Jastrow et al. 2007). Furthermore, the lack of nutrients and energy sources may restrict the growth of microorganisms involved in decomposition processes.

The amount of C stored in the soil is controlled by the potential of soils to stabilize organic C. Various studies have shown that less CO_2 is released from soils where large amounts of C are associated with the mineral phase (Six et al. 2002). This is generally explained by an interaction between reactive mineral surfaces and organic C by mechanisms such as ligand exchange, polyvalent cation bridges, and hydrophobic interactions (Vogel et al. 2015; von Lutzow et al. 2006). Therefore, the size of the clay and silt fraction is important for the storage of soil C (Tan et al. 2004). Moreover, chemical soil properties such as the pH value or the cation exchange capacity can affect the microbial turnover of organic matter (Ladegaard-Pedersen et al. 2005; Leuschner et al. 2013).

Direct and indirect anthropogenic activities influence the C dynamic in forest ecosystems. Various studies showed an impact of tree species selection and accompanied forest stand composition on C stocks (Ladegaard-Pedersen et al. 2005; Prietzel and Bachmann 2012). Forest management practices such as drainage, thinning, timber harvesting, and liming can also affect soil C dynamics (Johnson et al. 2002; Nave et al. 2010).

Investigations of liming induced effects on C stock changes have revealed contradictory results. On the one hand, it has been shown that after the application of lime, the decomposition of the organic layer increased accompanied by an accumulation of C in the mineral soil (Andersson and Nilsson 2001; Evers et al. 2008). On the other hand, various studies have demonstrated that a long-lasting decrease ranging from weeks up to 1 year followed the initial increase in soil respiration after liming (Illmer and Schinner 1991; Melvin et al. 2013).

In addition, various studies suggested an impact of nitrogen (N) deposition on C sequestration in forest soils, but the results are highly uncertain and vary by two orders of magnitude (de Vries et al. 2009; Janssens et al. 2010). The increase in N

deposition on forests over a longer time period may reduce the decomposition of organic matter. Increased organic matter input in soil through enhanced above-ground biomass productivity or increased recalcitrance of N-enriched litter may lead to reduced long-term decomposition rates of organic matter (de Vries et al. 2009).

Results of the 1st National Forest Inventory (NFSI) were presented by Wolff and Riek (1996), who showed spatial patterns of C stocks and morphological humus forms and found correlations to bedrock, soil texture, and main soil types. They reported effects of a partial decoupling of material cycles on C dynamics due to long-lasting acid and N deposits during the last decades. Estimates of nationwide C pools and repeated soil C inventories or soil C monitoring are still rare, even though such information is essential to fulfill national commitments under Article 3.4 of the Kyoto Protocol and under the 2016 Paris climate agreement. With the repetition of the NFSI, Germany obtained a comprehensive data base to evaluate the status quo of soil organic matter and the C sequestration rate of forested soils over time.

6.2   Carbon Stocks in German Forest Soils

6.2.1   Carbon Stocks

Forest soils (the organic layer and the mineral soil to a maximum depth of 90 cm) contained on average 117.1 ± 1.7 Mg ha^{-1} of organic C. The median of the organic C stock was 105.0 Mg ha^{-1} indicating a right-skewness with a disproportionately large number of sites with high organic C stocks (Fig. 6.1). The 10% soils with the lowest organic C stocks contained less than 64.8 Mg ha^{-1}, while the 10% soils with the highest organic C stocks contained more than 177.4 Mg ha^{-1}. Highest organic C stocks were usually found in soils where the decomposition of organic matter was restricted due to oxygen deficiency caused by water saturation and/or by per humid climate. Soils that are limited in depth by continuous rocks, gravel, or stones contained the lowest organic C stocks. In respect to the amount of C stored in the whole soil, the organic layer contained 18% of C (Fig. 6.2a). Organic C stocks in the organic layer were highly variable with C stocks ranging from 0.2 to 190 Mg ha^{-1} and a coefficient of variation of 94%. The upper 30 cm of the mineral soil contained 59% of the organic C stored in the complete profile, while the depth increments 30–60 cm and 60–90 cm contained 18% and 9% of the total organic C, respectively. This gradient is similarly to observe N stocks (see Chap. 5). In contrast to averaged C stocks which decreased with depth, the relative variability of C stocks increased with depth with a coefficient of variation ranging from 50% in depth increment 0–5 cm to 169% in depth increment 60–90 cm. The organic C stocks in different depth increments of the mineral soil varied considerably ranging from 0.4 to 142.5 Mg ha^{-1} in depth increment 0–5 cm and from 0.6 to 590.9 Mg ha^{-1} in depth increment 10–30 cm.
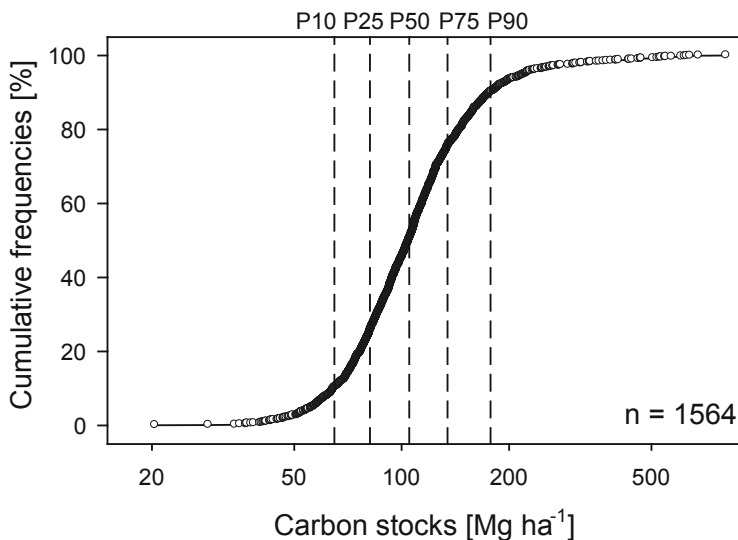
Fig. 6.1 Cumulative frequencies [in %] of organic stocks in German soils (organic layer and mineral soil down to a depth of 90 cm). 10, 25, 50, 75, and 90% quantiles are indicated (P10, P25, P50, P75, P90)

6.2.2 Organic Carbon Stock Changes in German Forest Soils

Estimates of organic C stock changes in the organic layer and mineral soil between the inventories are based on sites that were sampled at both inventories. Bogs and fens were not included in the analyses. The thickness of the humus body in wetland soils varies over time, and therefore, a reference level for a repeated measurement is nonexistent. Carbon stocks of the organic layer increased negligible by 0.06 ± 0.03 Mg ha^{-1} year^{-1} since the first inventory (Fig. 6.2b). In contrast, C stocks increased significantly in increments up to a depth of 30 cm with 0.25 ± 0.02 Mg ha^{-1} in 0–5 cm, 0.16 ± 0.01 Mg ha^{-1} in 5–10 cm, and 0.28 ± 0.03 Mg ha^{-1} in 10–30 cm ($p < 0.001$). Significant organic C stock changes in the soil below 30 cm could not be found. The averaged organic C stock in German forest soils down to a depth of 90 cm increased significantly by 0.75 ± 0.09 Mg ha^{-1} year^{-1} resulting in a total increase of 11.3 Mg ha^{-1} between the inventories (a time period of 15 years). In a previous analysis, the organic layer was identified as a stable C pool, while the mineral soil sequestered 0.41 Mg ha^{-1} year^{-1} (Grüneberg et al. 2014). The apparent tendency of an accumulation in the organic layer and the upper depth increment of the mineral soil are also reflected by N stocks (see Chap. 5). The pattern of decreasing N stocks with depth is mainly due to high differences in 30–60 cm which was not observed for C. Since the measured concentrations are much higher for C compared to N, they are less affected by potential analytical errors. On the European scale estimates of C stock changes, rates ranged from 0.1 to 1.0 Mg ha^{-1} year^{-1} (Luyssaert et al. 2010). Jonard et al. (2017) quantified changes in soil organic C stocks based on the data from two soil
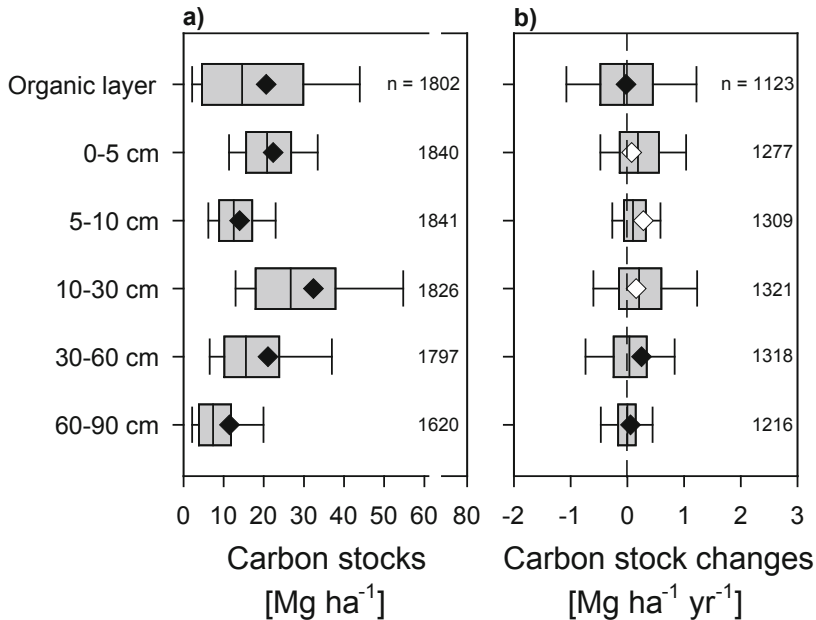
Fig. 6.2 (**a**) Carbon stocks and (**b**) carbon stock changes in the organic layer and depth increments of the mineral soil. Box plots on the carbon stocks refer to all plots considered in the 2nd National Forest Soil Inventory, while carbon stock changes refer to plots exclusively considered on both National Forest Soil Inventories. Box plots with white diamonds on carbon stock changes indicate significant differences based on a one-sample t- test with H0: 0 ($p < 0.05$). Organic soils were excluded from the analysis of carbon stock changes

surveys carried out in the French forest monitoring plots. The authors estimated for forest soils across France an accumulation of 0.35 Mg C ha^{-1} year^{-1} between 1993 and 2012.

For the first time, Grüneberg et al. (2014) estimated countrywide C stock changes of forest soils in Germany. Differences between these results and results presented here arise from different statistical approaches. In contrast to the results presented here which are based on paired samples, previous estimates of C stocks and C stock changes considered both paired and unpaired samples. They applied site-specific forest stand and soil information which was available from nationwide geographical datasets. The plots were grouped into classes, which correspond to the classification scheme of the geographical datasets. For the organic layer, inventory plots were grouped into classes of forest stand types (coniferous forest, mixed forest, and broadleaf forest) and substrate groups (dystrophic and eutrophic unconsolidated sediments as well as rocks with high and low base saturation). For the mineral soil, inventory plots were grouped according to soil and substrate information. Federal states were included in the classification to consider differences in the sampling density among German federal states. Detailed information about the methods can be found in Grüneberg et al. (2014).
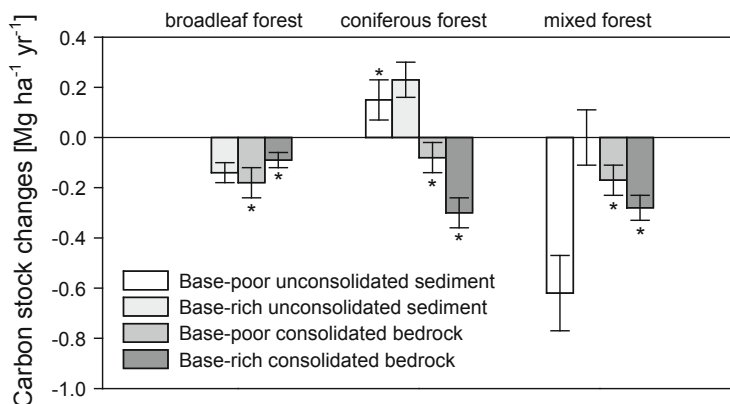
Fig. 6.3 Organic carbon stock changes in the organic layers of different substrate groups under various forest stands between the 1st and 2nd National Forest Inventory (Grüneberg et al. 2014). Asterisks indicate a 0.05 significance level, tested by a two-way ANOVA and the attended Tukey HSD test for each substrate group individually

The C stock changes of the organic layer ranged from $-0.62 \pm 0.15 \, \text{Mg ha}^{-1} \, \text{year}^{-1}$ under broadleaf forest to $0.23 \pm 0.07 \, \text{Mg ha}^{-1} \, \text{year}^{-1}$ under coniferous forest, both in soils derived from eutrophic unconsolidated sediments (Fig. 6.3). The site-specific organic C stocks have increased since the first national inventory especially in the North German Lowland and in the Alpine foothills (Fig. 6.4). Both regions are characterized by dystrophic unconsolidated sediments and coniferous forests. In contrast, C was not sequestered but rather released under broadleaf and mixed forest. In the low mountain ranges consisting of solid rocks with low base saturation in the middle and southern parts of Germany, the organic layer is considered to be a C source as it released C. Carbon was also released from organic layers in the Alps as well as in the mountainous regions of southern Germany with solid rocks high in base saturation. The mountainous regions with high base saturation are not characterized by specific forest stands. However, organic layers in broadleaf and mixed forest lost more C than in coniferous forest.

The C stock changes in the upper 30 cm of the mineral soil ranged from $-0.71 \pm 0.28 \, \text{Mg ha}^{-1} \, \text{year}^{-1}$ in soils derived from redeposited material from lime stone, marlstone, and dolomite to $1.35 \pm 0.19 \, \text{Mg ha}^{-1} \, \text{year}^{-1}$ in soils derived from eutrophic sandy deposits (Fig. 6.5). Similar to the organic layer, the mineral soil to a depth of 30 cm showed a strong increase in C stocks in mostly sandy soils of the North German Lowland (Fig. 6.6). The largest changes could be detected in the eastern part of the North German Lowland, where the initial C pool was low. Similar increases could be detected in the northern part of the Upper Rhine Valley and in the Rhine-Main Lowland. Moderate negative changes were found in soils derived from solid rocks of mountains and hills especially in the middle and south of Germany as well as in soils of the Alps. Furthermore, soils derived from limestone weathering or from weathered marlstone and claystone as, for example, in the Swabian and Franconian Alb and Upper Palatinate Jura showed slightly decreasing C stocks.
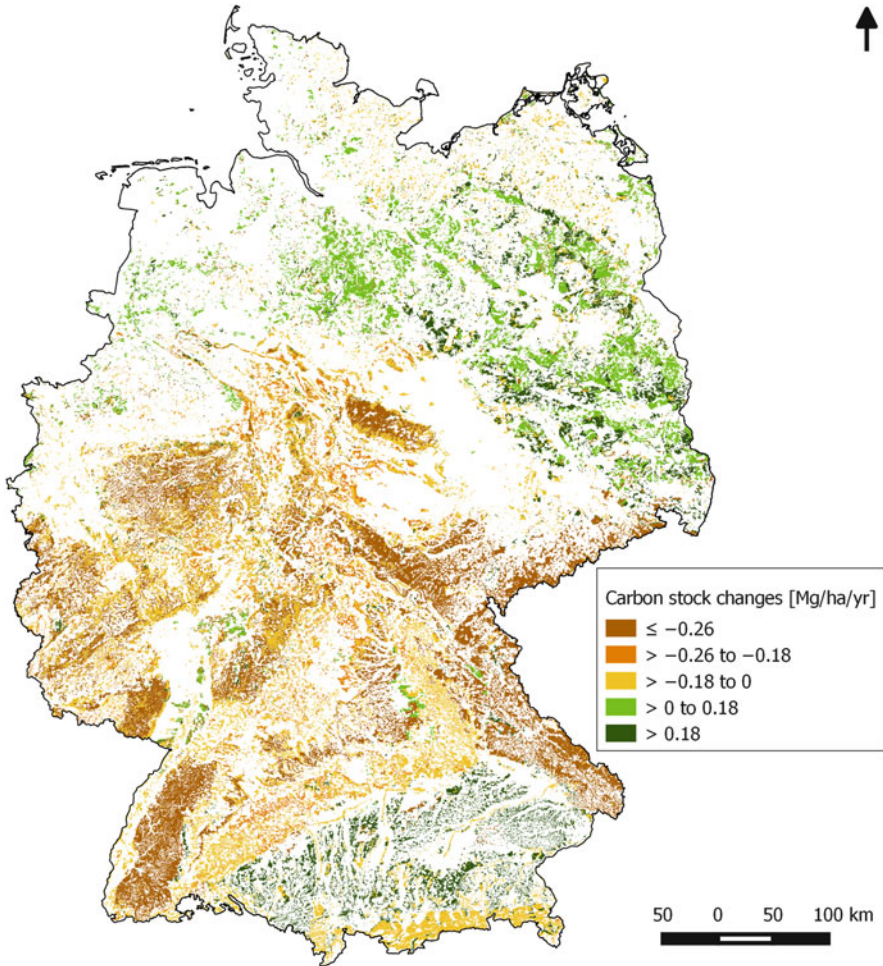
Fig. 6.4 Area-related organic carbon stock changes in the organic layer of German forest soils between the 1st and 2nd National Forest Soil Inventory

6.2.3   Effects of Forest Stands and Parent Material on Carbon Stocks

6.2.3.1   Forest Stands-Specific Carbon Stocks

Effects of forest stands on C stocks of the organic layer and mineral soil were investigated by analyses of variance (ANOVA) and a subsequent pairwise comparison between the groups by Bonferroni test ($p < 0.05$). Forest stands were distinguished in oak, beech, other broadleaf stands, pine, spruce, other coniferous stands, and mixed stands. In the organic layer, C stocks ranged from 7.8 ± 1.1 Mg ha^{-1}
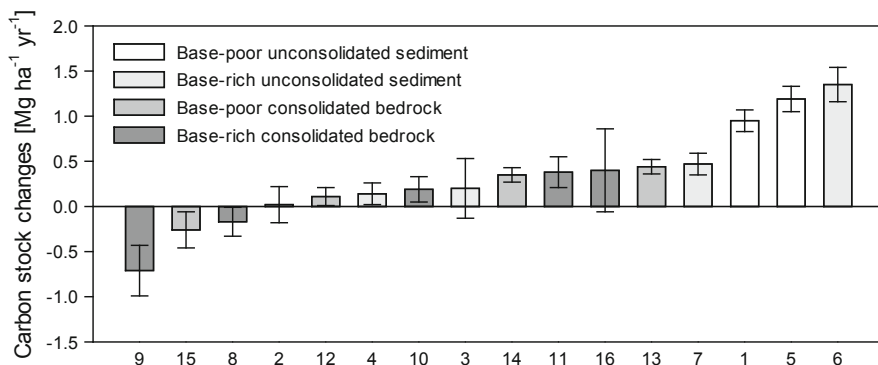
Fig. 6.5 Organic carbon stock changes in the mineral soil down to a depth of 30 cm under different soil and substrate groups between the 1st and 2nd National Forest Soil Inventory. Numbers on the *X*-axis represent soil parent material groups defined in Grüneberg et al. (2014)

under other broadleaf trees to 31.4 ± 0.9 Mg ha^{-1} under pine (Fig. 6.7a). They differed significantly among forest stands (ANOVA; $p < 0.001$). Apart from the comparison between beech with oak and between other coniferous trees with mixed stands, the C stocks between all other stands were significantly different. Broadleaf forest stands showed the lowest values, while C stocks of coniferous forest stands were an order of magnitude higher. This can be explained by an inhibited decomposition of needle litter and the associated enrichment of organic matter in the organic layer.

Broadleaf forest stands stored with 88.9 ± 5.0 Mg ha^{-1} in the upper 30 cm of the mineral soil the highest amount of organic C, while pine forest stands showed with 60.0 ± 1.9 Mg ha^{-1} the lowest stocks (Fig. 6.7b). The organic C stocks in the mineral soil to a depth of 30 cm were, though less pronounced, still significantly different between the forest stands (ANOVA; $p < 0.001$). Significantly higher C stocks were found under other broadleaf forest than under other forest stands, while pine forest stands stored significantly less C in this depth increment than other forest stands.

A significant part of the C is observed in 30–90 cm emphasizing the subsoil as a stable, long-term C pool. The amount of C stored in the subsoil was half the C stored in the 0–30 cm depth increment (Fig. 6.7c). The variation of C stocks among the groups in the subsoil (30–90 cm) was lower than in the depth increment above. Nevertheless, significant differences among the forest stands were evident (ANOVA; $p < 0.001$). The variation within the groups, however, was similar to the 0–30 cm depth increment. Broadleaf forest stands are prone to store more C in the subsoil than coniferous forest. Under pine forest stands, we detected with 24.8 ± 1.7 Mg ha^{-1} the lowest C stocks. Similar to the 0–30 cm depth increments, other broadleaf forest stands stored significantly higher C (42.6 ± 4.0 Mg ha^{-1}) in the subsoil than all other forest stands. At the same time, also the largest variation was found under other broadleaf forest stands probably due to a high variability of site conditions among the stands. However, the pattern that more C is stored under
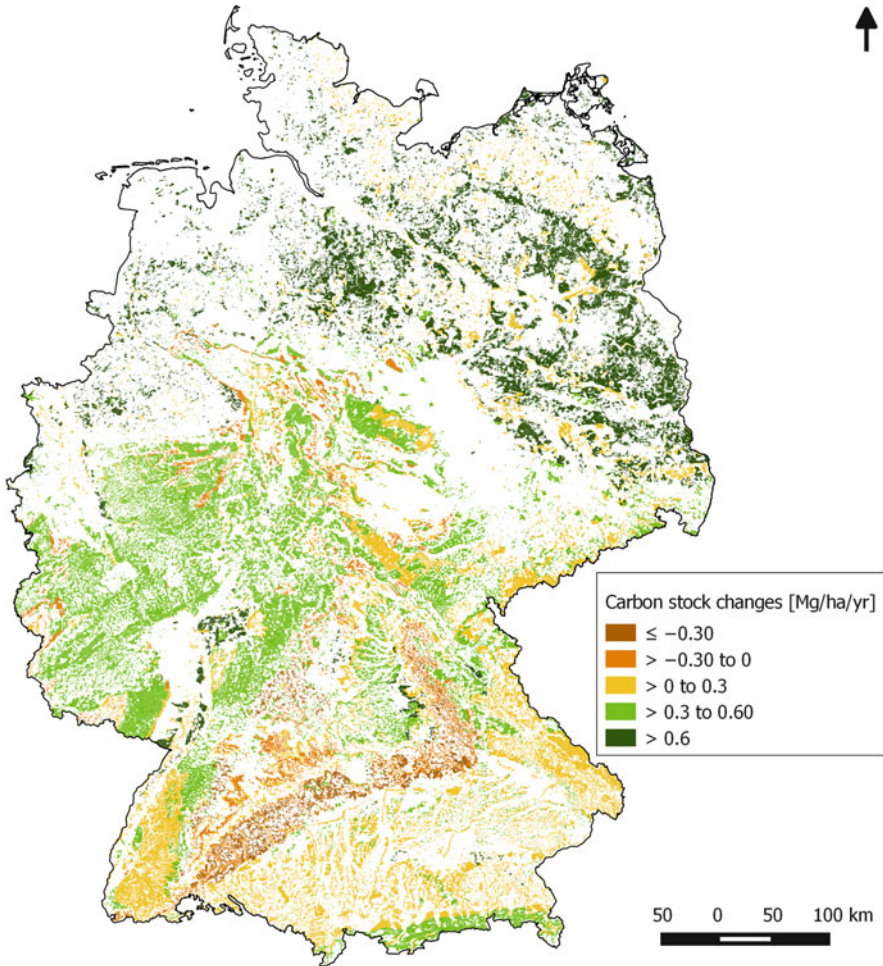
Fig. 6.6 Area-related organic carbon stock changes in the mineral soil down to a depth of 30 cm of German forest soils between the 1st and 2nd National Forest Soil Inventory

broadleaf compared to coniferous forest stands might be caused by regional and site-specific differences of soil properties.

6.2.3.2 Organic Carbon Stocks of Different Soil Parent Materials

The soil parent material groups determine to a large extend soil physical and chemical properties of a forest site that in turn affect the amount and turnover of soil organic matter. Differences of organic C stocks among soil parent material groups were analyzed by ANOVA and subsequent pairwise comparisons between
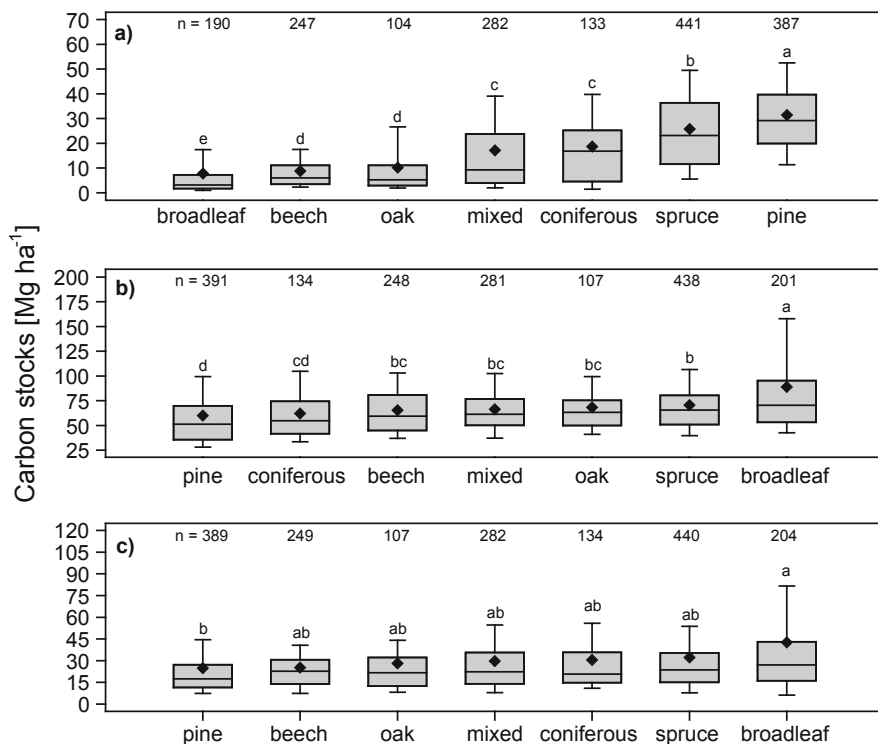
Fig. 6.7 Organic carbon stocks in (**a**) the organic layer, (**b**) the mineral soil down to a depth of 30 cm and (**c**) in the depth increment 30–90 cm under various forest types. Forest types with different letters are significantly different ($p < 0.05$). The line within a box represents the median and the diamond represents the mean. Forest types on the *X*-axis; for definition see Chap. 2

the groups by Bonferroni test ($p < 0.05$). Soils of bogs, fens, and other soils with high organic C concentrations (>15%) as well as soils from the Alps were not considered in the ANOVA. Soils of the Alps were excluded because of their low sample size, while organic soils were not studied due to their clearly distinctive development, properties, and reaction to environmental effects compared to mineral soils. In particular the thickness of the peat layer determines the amount of organic C stored in organic soils. Peat layers are characterized by notably high C stocks and a high spatial variability of organic C stocks. The classification into different soil parent material groups, however, could reduce the variability within the groups compared to the forest stand types because highly variable organic soils were analyzed separately as organic soils.

The results showed a relation of C stocks between the organic layer and the parent material. The averaged C stored of the organic layer ranged from 6.2 ± 0.5 Mg ha^{-1} in soils from weathered carbonate bedrock to 29.6 ± 1.1 Mg ha^{-1} in soils from base-poor unconsolidated sediment (Fig. 6.8a). Forest sites rich in nutrients and base cations are characterized by low organic C stocks in the organic layer due to favorable
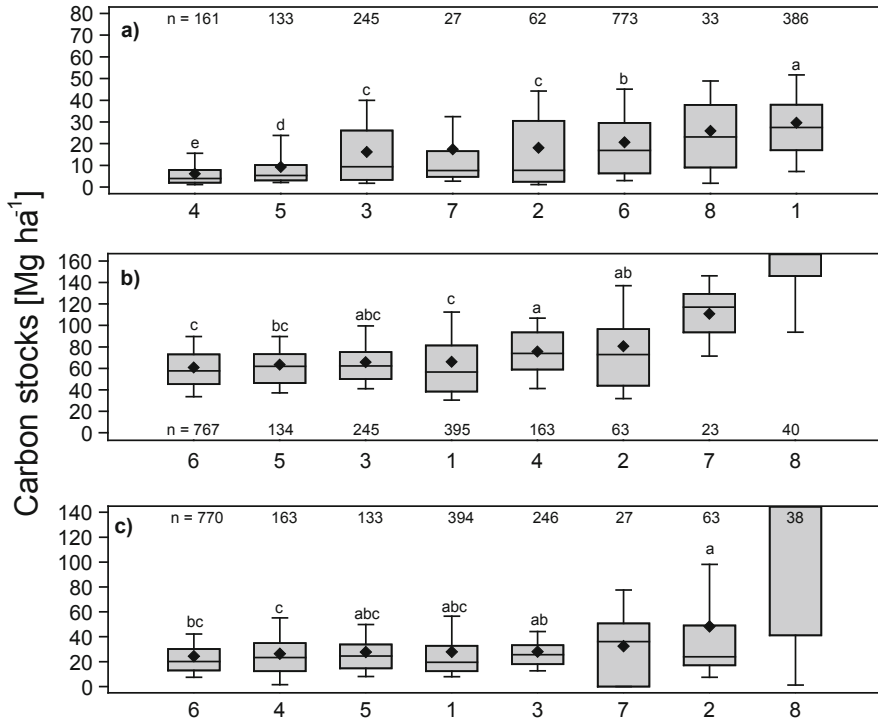
Fig. 6.8 Organic carbon stocks in (**a**) the organic layer, (**b**) the mineral soil down to a depth of 30 cm, and (**c**) the depth increment 30–90 cm in soils derived from various parent materials. Soil parent material groups with different letters are significantly different ($p < 0.05$). The line within a box represents the median, and the diamond represents the mean; numbers on the X-axis represent soil parent material groups with (1) soils from base-poor unconsolidated sediment, (2) soils of alluvial plains, (3) loamy soils of the lowland, (4) soils from weathered carbonate bedrock, (5) soils from basic-intermediate bedrock, (6) soils from base-poor hard bedrock, (7) soils from the Alps, and (8) organic soils

conditions for soil organisms to decompose organic matter. In contrast, the high organic C stocks in soils from base-poor unconsolidated sediment or in organic soils can be explained through reduced decomposition of organic matter due to low nutrition availability or anaerobic conditions. The Bonferroni test revealed significantly higher C stocks of the organic layer in soils from base-poor unconsolidated sediment than in all other groups. The organic layer of soils from base-poor bedrock stored significantly more C than the organic layer of soils from weathered carbonate bedrock, soil from basic-intermediate bedrock, as well as loamy soils of the lowland and soils in broad river valleys. Carbon stocks were also significantly higher in the organic layer of soils in broad river valleys and of loamy soils of the lowland compared to the organic layer of soils from weathered carbonate bedrock and of soils from basic to intermediate bedrock.

Organic layer C stocks were stronger influenced by forest stand types than by soil parent material, while C stocks of the mineral soil down to 30 cm depth were most affected by soil parent material. Soil C stocks were higher and characterized by a considerably larger variability than C stocks in the organic layer (Fig. 6.8b). Moreover, the magnitude of organic C stored in the mineral soil derived from different parent materials was contrary to the amount of C stored in the organic layer. Base cation-poor sites showed lower organic C stocks in the mineral soil than sites rich in base cations. The averaged values for the 0–30 cm depth increment ranged between 60.9 ± 0.9 t C ha^{-1} in soils from base-poor bedrock and 110.9 ± 5.7 t C ha^{-1} in soils from the Alps. The high organic C stocks in the soils from the Alps can be explained by a slow decomposition of organic matter due to effects of high altitude. The studied organic soils stored with 216.6 ± 19.0 Mg ha^{-1} much more C than mineral soils. Soils in broad river valleys and soils from weathered carbonate bedrock were significantly more enriched in C than soils from base-poor unconsolidated sediment and bedrock. Particularly soils in broad river valleys received organic matter by lateral transport during past flooding events.

Carbon stocks of the subsoil (depth increment 30–90 cm) hardly differed among the soil parent materials. Moreover, the variability within most of the groups was less pronounced compared to the 0–30 cm depth increment. The average organic C stocks ranged between 24.3 ± 0.8 Mg ha^{-1} in soils from base-poor bedrock and 48.2 ± 8.8 Mg ha^{-1} in soils in broad river valleys (Fig. 6.8c). Organic soils exhibited the highest organic C stocks in 30–90 cm (179.3 ± 24.8 Mg ha^{-1}). In contrast to the 0–30 cm depth increment, the C stocks in soils in broad river valleys were significantly higher than in soils from weathered carbonate bedrock. The differences between the 0–30 cm and 30–90 cm depth increments might be explained by higher stone contents in sites derived from carbonate rocks compared to sites derived from material transported by water as in broad river valleys. The organic C stocks of loamy soils of the lowland or of soils from basic-intermediate bedrock were not different from sites poor in cations although the mineralogical composition of those soils was expected to stabilize larger amounts of C over a long period of time.

6.2.3.3 Interactions Between Forest Stand Types and Soil Parent Material

One-dimensional considerations prevent separating interactions between forest stand types and soil parent material due to the low sample size and the high variability of any of the groups. Coniferous and broadleaf forest stands were separated insufficiently by the NFSI method because pure stands were separated in which at least 70% of the trees in the main canopy are of a single species. This means that pure forest stands can provide 30% of other tree species. To enhance the number of sample points for a sufficient statistical analysis in each class, we further distinguished between pure coniferous and broadleaf forest stands based on a proportion of 90% of single tree species in the main canopy. The following soil parent material groups were considered for the analyses: soils from base-poor consolidated bedrock,
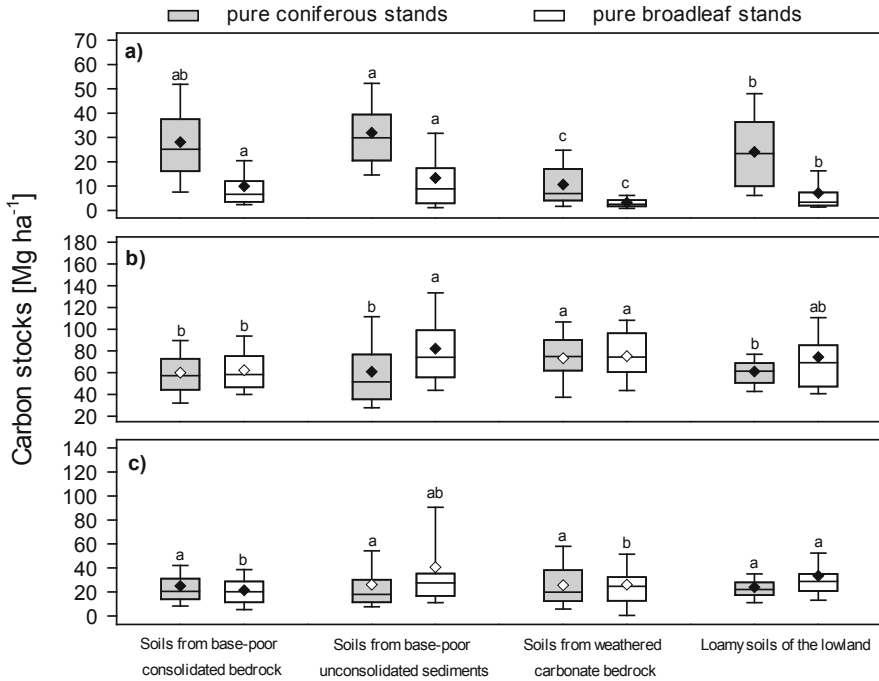
Fig. 6.9 Carbon stocks in the organic layer (**a**), as well as in the mineral soil down to a depth of 30 cm (**b**) and in depth increment 30–90 cm (**c**) under pure coniferous and broadleaf stands on soils derived from various parent materials. Black diamonds indicate significant differences between pure stands within the same soil parent material group based on a two-sample t-test ($p < 0.05$). Comparisons of the same pure stand among soil parent material groups are based on Bonferroni test ($p < 0.05$). Same letters indicate no significant differences ($p < 0.05$). Outliers are masked; a line within a box represents the median, and the diamond represents the mean

Table 6.1 Results of analysis of variance (ANOVA) testing effects of pure forest stands and soil parent material on carbon stocks in the organic layer

Variables	DF	MSE (type III)	F value	p value
Pure forest stand	1	260.965	371.43	<0.0001
Soil parent material	3	112.663	53.45	<0.0001
Pure forest stand*soil parent material	3	4.788	2.27	0.0786

DF degrees of freedom, *MSE* mean square error

soils from base-poor unconsolidated sediments, loamy soils of the lowland, and soils from weathered carbonate bedrock.

The organic C stocks of the organic layer ranged from 3.2 ± 0.3 Mg ha^{-1} under pure broadleaf forest in soils from weathered carbonate bedrock to 31.9 ± 1.04 Mg ha^{-1} under pure coniferous forest in soils from base-poor unconsolidated sediments (Fig. 6.9a). The ANOVA revealed a significant relation between organic C stocks in the organic layer of pure forest stands and soil parent material group (Table 6.1).

Table 6.2 Results of analysis of variance (ANOVA) testing effects of pure forest stands and soil parent material on carbon stocks in the mineral soil down to a depth of 30 cm

Variables	DF	MSE (type III)	F value	p value
Pure forest stand	1	2.729	14.93	0.0001
Soil parent material	3	4.171	7.61	<0.0001
Pure forest stand*soil parent material	3	2.323	4.23	0.0055

DF degrees of freedom, *MSE* mean square error

Similar to the classification into soil parent material groups without distinguishing forest types among forest stands, where coniferous forest showed the highest and broadleaf forest showed the lowest C stocks, here we found maximum organic C stocks under pure coniferous forest and minimum organic C stocks under pure broadleaf forest across all soil parent material groups. The differences between the pure forest stands were significant for all soil parent material groups. We found significant differences under pure coniferous forest exclusively between soils from base-poor consolidated bedrock and loamy soils of the lowland and soils from base-poor unconsolidated sediments, respectively. Taking broadleaf forest into account, significant differences were shown between most of the groups except for comparing soils from base-poor unconsolidated sediments with soils from base-poor consolidated bedrock. It could be shown that organic C stocks in organic layers are affected by soil parent material as we found higher organic C stocks on base cation-poor substrates compared to substrates low in base cations where in turn the organic C stocks were lower.

For average C stocks in the 0–30 cm depth increment, the variability within the groups was much larger than between the groups. The organic C stocks reached from 60.2 ± 1.3 t C ha^{-1} in soils from base-poor consolidated bedrock under coniferous forest to 84.9 ± 7.3 t C ha^{-1} in soils from base-poor unconsolidated sediments under broadleaf forest (Fig. 6.9b). In contrast to the organic layer, the ANOVA revealed significant higher C stocks under broadleaf than under coniferous forest (Table 6.2). Taking the soil parent material into account, soils from base-poor unconsolidated sediments and loamy soils of the lowland showed significant higher C stocks under broadleaf forest in comparison with coniferous forest. Coniferous and broadleaf forest stored significantly more C in soils from weathered carbonate bedrock than in soils from base-poor consolidated bedrock. Under coniferous forest C stocks were significantly higher in soils from weathered carbonate bedrock compared to soils from base-poor unconsolidated sediments and in loamy soils of the lowland. Furthermore, under broadleaf forest, the amount of C stored in soils from base-poor unconsolidated sediments was significantly higher than in soils from base-poor consolidated bedrock. The results of the ANOVA showed, however, significant interactions between pure stands and soil parent groups.

In the 30–90 cm depth increment (subsoil), we estimated averaged C stocks ranging from 22.4 ± 1.2 Mg ha^{-1} in soils from base-poor consolidated bedrock to 37.8 ± 6.5 Mg ha^{-1} in soils from base-poor unconsolidated sediments (Fig. 6.9c). Both the highest and lowest C stocks were found under broadleaf forest regardless of the

Table 6.3 Results of analysis of variance (ANOVA) testing effects of pure forest stands and soil parent material on organic carbon stocks in the mineral soil depth increment 30–90 cm

Variables	DF	MSE (type III)	F value	p value
Pure forest stand	1	4.524	8.67	0.0033
Soil parent material	3	9.521	6.08	0.0004
Pure forest stand*soil parent material	3	11.420	7.30	<0.0001

DF degrees of freedom, *MSE* mean square error

soil parent material. The C stock variability in the 30–90 cm depth increment is similar to that in the 0–30 cm depth increment though the C stored in the subsoil was half the C stored in the mineral soil down to 30 cm depth. The ANOVA revealed a significant relation to the forest type for the 30–90 cm depth increment (Table 6.3). The C stored in loamy soils from the lowland was significantly higher under broadleaf forest compared to coniferous forest. In contrast, soils from base-poor consolidated bedrock showed under coniferous forest higher C stocks than under broadleaf forest. Nevertheless, C stocks under broadleaf forest in the subsoil were prone to be affected by soil parent material. This can be proved by loamy soils of the lowlands which stored significant more C than soils from base-poor consolidated bedrock or soils from weathered carbonate bedrock. In contrast to broadleaf forest, the C stocks among the soil parent material groups were not significantly different. Similar to the 0–30 cm depth increment, the ANOVA revealed significant interactions between forest stands and soil parent material groups.

6.3 Effects of Natural and Anthropogenic Environmental Factors on Carbon Stocks in Forest Soils

We used structural equation modelling (SEM) to analyze direct and indirect factors affecting organic C stocks and organic C stock changes. Structural equation modelling was performed with R 3.2.2 (R Core Team 2015) by means of the work package lavaan (Rosseel 2012). A hypothetical model was created initially before pre-postulated cause-effect relationships could be verified by the data of the NFSI. Structural equation models can be used to analyze simultaneously dependent and independent variables in order to study more complex relations among variables. Independent variables are denoted as exogenous, and dependent variables are denoted as endogenous. Results of the SEM can be represented graphically by path analyses where the strength of the causal relationship is expressed by path coefficients. A direct effect revealed by the SEM and showed by the path analyses is presented with an aligned arrow, while covariance is displayed with a curved double arrow. It is supposed that on national scale the amount of C stored in the organic layer and mineral soil is mainly affected by climate, plant biomass, and related litter inputs as well as physical and chemical soil properties. Further effects can arise from forest management practices and deposition of N. For the modelled effects on
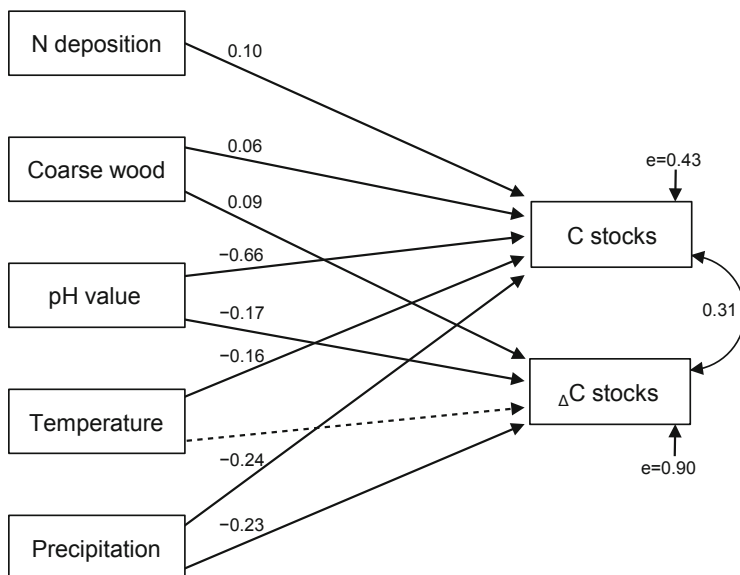
Fig. 6.10 Results of the structural equation modelling with $n = 1114$, $P(\chi 2) = 0.831$, 11 degrees of freedom, and a RMSEA (root mean square error of approximation) of 0 (90% confidence interval, [0; 0.047]) for the organic layer with exogenous (coarse wood = coarse wood volume, pH value [1 M KCl], temperature = mean annual temperature, precipitation = mean annual precipitation) and endogenous (C stocks = organic carbon stocks, ΔC stocks = annual organic carbon stock changes between NFSI I and NFSI II) variables in boxes and path coefficients with significant (continuous arrow) and not significant (dashed arrow) assignments as well as error variances (e)

organic layer C, we considered the averaged mean temperature and the annual precipitation, the coarse wood volume per hectare, the averaged mean N deposition, as well as the pH value (1 M KCl) as variables describing site properties. The more complex model for the mineral soil down to 30 cm depth additionally included the clay concentrations as an exogenous variable and the cation exchange capacity and the C concentration as endogenous variables. It should be considered here that the effective cation exchange capacity (CEC) is partially determined by the C concentration.

The results of the SEM for the organic layer showed that the magnitude of C stocks was strongest correlated to the pH value (−0.66), while relations to the climate variables temperature (−0.16) and precipitation (−0.24) were less distinct, and relations to the N deposition (0.10) as well as to the coarse wood volume (0.06) were weak (Fig. 6.10). Carbon stock changes in the organic layer were affected by coarse wood volume (0.09), pH value (−0.17), and precipitation (−0.23). The model for the mineral soil revealed weak correlations between organic C stocks and the exogenous variables N deposition (0.18) and pH value (0.06) (Fig. 6.10). In contrast, organic C stocks were strongly affected by the endogenous variable organic C concentration which in turn was largely determined by the clay concentrations (0.48). The C concentration was not considered in the model for the organic layer
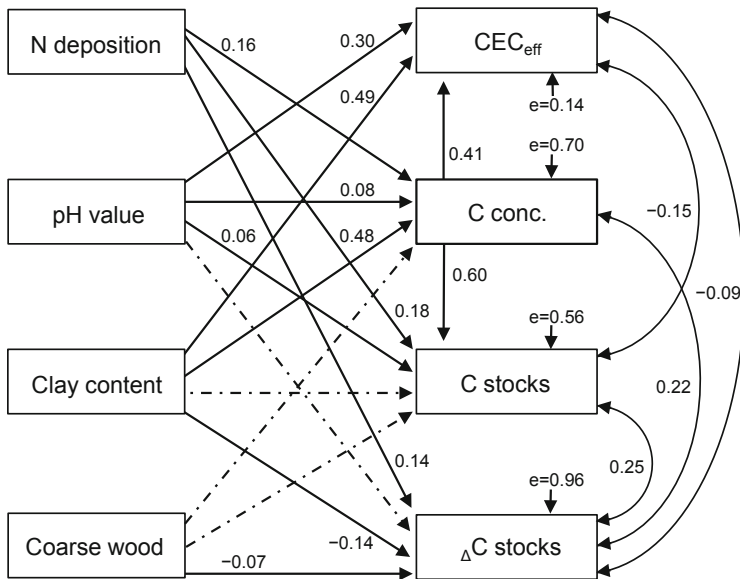
Fig. 6.11 Results of the structural equation modelling with $n = 886$, $P(\chi 2) = 0.330$, 22 degrees of freedom, and a RMSEA (root mean square error of approximation) of 0.011 (90% confidence interval, [0; 0.069]) for the mineral soil down to a depth of 30 cm with exogenous (coarse wood = coarse wood volume, clay concentrations, pH value [1 M KCl], N deposition = mean annual N deposition) and endogenous (CEC$_{eff}$ = effective cation exchange capacity, C conc. = organic carbon concentrations, C stocks = organic carbon stocks, $_\Delta$C stocks = annual organic carbon stock changes between NFSI I and NFSI II) variables in boxes and path coefficients with significant (continuous arrow) and not significant (dashed arrow) assignments as well as error variances (e)

because the amount of C stored in the organic layer is mainly controlled by the thickness and mass of the organic layers and not—as in the mineral soil—controlled by the C concentration. Additionally to direct effects on organic C stocks, coefficients revealed a path of N deposition (0.16) and pH value (0.08) on C concentration. Indirect effects on organic C stocks—that were weaker than direct effects—operating through C concentration were therefore caused by N deposition (0.10) and pH value (0.05). The clay concentrations indirectly affected (0.32) C stocks. The indirect effect was much larger than the direct effect, which was not significant disguised the negative relation of the direct effect between clay concentrations and C stocks.

The SEM revealed that the C stock changes of the mineral soil depend on N deposition (0.14), clay concentrations (−0.14), and coarse wood volume (−0.07) (Fig. 6.11). Additionally, a covariance (0.25) was shown between C stocks and C stock changes. Taking this covariance into account, N deposition (0.05) operated through C stocks on C stock changes. Further relations with C stock changes resulted from indirect effects of the exogenous variables N deposition (0.04), pH value (0.02), and clay concentrations (0.11) operating through the C concentrations and

the respective covariance. Furthermore, clay concentrations (0.05) were also related with C stock changes through the covariance of the effective cation exchange capacity. All revealed effects operating through the covariance are less pronounced than the revealed direct effects. In both models the magnitudes of the sequestered C strongly depend on the amount of C stored in the soil. Various studies demonstrated negative relationships between initial soil C stocks and C stock changes which were explained by a change in the forest management (Goidts and van Wesemael 2007; Riley and Bakkegard 2006) or by changed environmental conditions (Bellamy et al. 2005; Saby et al. 2008). Repeated measurements on the same subject or unit of observation can lead to a statistical phenomenon called "regression to the mean" (Barnett et al. 2005). This statistical effect has been debated causing artifact losses from initially C-rich soil (Lark et al. 2006) and C gains in soils with initially low C stocks (Callesen et al. 2015). The revealed covariance between C stocks and C stock changes indicated the effect of the regression to the mean.

6.4　Effects of Natural Environmental Factors

The results of the studied soil parent material groups as well as the results of the SEM showed that site conditions are important factors affecting the storage of soil organic C. Base cation-poor sites showed for the organic layer highest C stocks, while base cation-rich sites stored low amounts of C (Fig. 6.12). Effects of soil parent material on the amount of stored C in the organic layer and mineral soil have been shown by various studies. It was demonstrated that the decomposition of litter was higher on sites rich in base cations or that the C accumulation of the organic layer was caused by reduced decomposition of the litter on sites low in base cations (Ladegaard-Pedersen et al. 2005; Vesterdal et al. 2008). In contrast to the organic layer, the magnitude of C stocks in mineral soils was higher on base cation-rich sites. Similar results were presented by other studies (Davis et al. 2004; Richardson and Stolt 2013; Conforti et al. 2016), while Bavarian forest soils showed no relation between soil parent material and C stocks in the mineral soil (Wiesmeier et al. 2013). The various groups of soil parent material are characterized by differing soil texture. It is well-known that clayey soils store more C than sandy soils. In the Good Practice Guidance for Land Use, Land-Use Change and Forestry (IPCC 2003), the texture is considered by default values for C stocks down to a depth of 30 cm for a warm temperate, moist climate. Sandy soils include all soils having >70% sand and <8% clay and are indicated to store 34 t C ha^{-1}. Soils with high activity clay minerals are lightly to moderately weathered soils, which are dominated by 2:1 silicate clay minerals. High activity soils are assumed to store 88 t C ha^{-1} (IPCC 2003). Texture-related differences in organic C stocks can be explained by a decrease in decomposing from sand to silt to clay-sized complexes. This is due to the positive relation between C concentrations and clay content, which may be attributed to the stabilization of organic matter by the formation of stable complexes with clay minerals (Six et al. 2002). The specific surface area of soils increases with decreasing
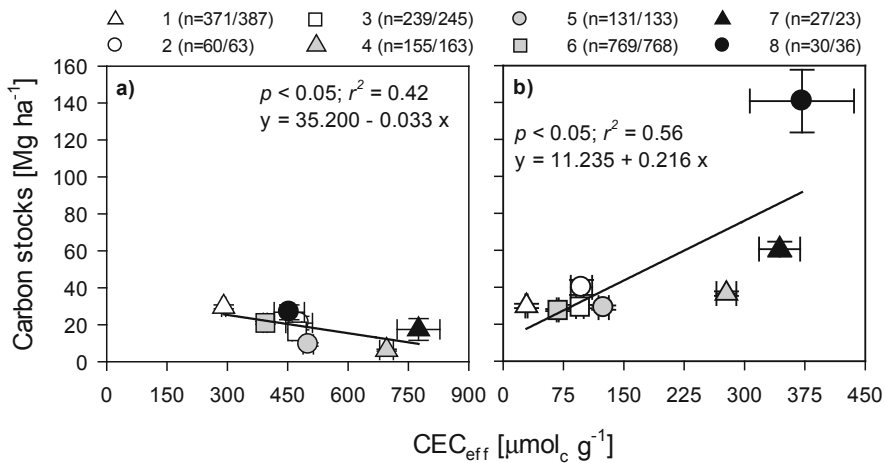
Fig. 6.12 Relation between the effective cation exchange capacity and carbon stocks (**a**) in the organic layer and (**b**) in the 10–30 cm depth increment of the mineral soil. The linear regression was performed between the averaged effective cation exchange capacity and carbon stocks of various soil parent material groups; the numbers in the legend represent soil parent material groups with (1) soils from base-poor unconsolidated sediment, (2) soils of alluvial plains, (3) loamy soils of the lowland, (4) soils from weathered carbonate bedrock, (5) soils from basic-intermediate bedrock, (6) soils from base-poor hard bedrock, (7) soils from the Alps, and (8) organic soils

particle size resulting in an increased importance of interaction between the organic matter and the mineral phase. Furthermore, an increased stability of soil organic matter might also be caused by its location within aggregates (Schrumpf et al. 2013; Golchin et al. 1997). The potential of a soil to form aggregates depends on the size distribution of the primary complexes and their characteristics which in turn depend on fundamental soil properties, such as clay mineralogy as well as on the abundance of microorganisms. Aggregates physically protect soil organic matter by forming physical barriers between microorganisms and their substrates and controlling food web interactions and consequently microbial turnover (Six et al. 2002). Moreover, the enhanced C concentration in fine-textured soils compared to soils with low clay content can probably be explained by higher soil moisture in clay-rich soils. Soils with high clay content have higher proportions of fine pores, which are less accessible for biological activities. We assume that this may have resulted in a reduced decomposition of the litter inputs at clay-rich sites and a subsequent accumulation of C (Grüneberg et al. 2013). In contrast to C stocks that correlate positively with clay content, we found increasing C sequestration with increasing particle size. The extent of decomposing soil organic matter decreases from sand- to silt- to clay-sized complexes suggesting that the binding capacity of clay and silt particles is more saturated in the finer-textured soils than in coarser- textured soils (Christensen 2001; Guggenberger et al. 1995). Schulten and Leinweber (2000) supposed that the importance of the clay fraction can become notably relevant with lower clay concentrations. Therefore, we assume that there is an enrichment of C in fine particle
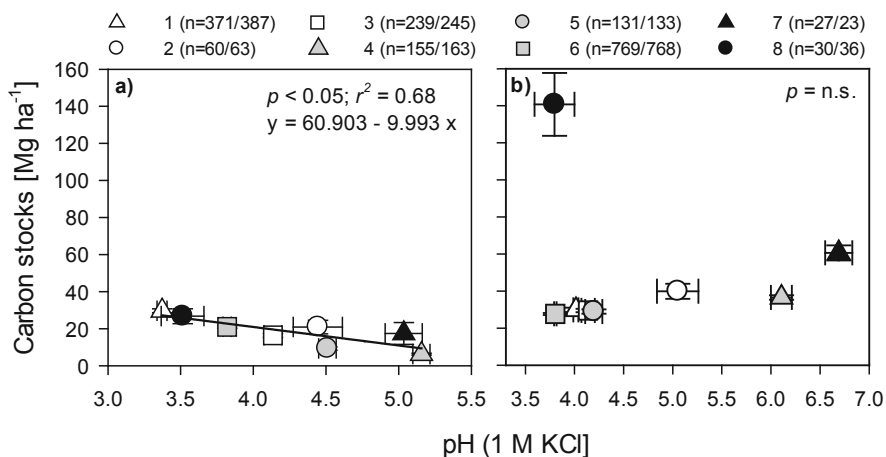
Fig. 6.13 Relation between the pH value and carbon stocks (**a**) in the organic layer and (**b**) in the 10–30 cm depth increment of the mineral soil. The linear regression was performed between the averaged pH value carbon stocks of various soil parent material groups; the numbers in the legend represent soil parent material groups with (1) soils from base-poor unconsolidated sediment, (2) soils of alluvial plains, (3) loamy soils of the lowland, (4) soils from weathered carbonate bedrock, (5) soils from basic-intermediate bedrock, (6) soils from base-poor hard bedrock, (7) soils from the Alps, and (8) organic soils

fractions in low clay sandy lowland soils. Consequently, the relative magnitude of C accumulation is elevated in low clay soils compared to high clay soils.

The SEM for the organic layer and the mineral soil revealed indications on a relation between soil acidity and C stocks. The pH value of the organic layer correlated negatively with C stocks, while the mineral soil showed a positive correlation. Furthermore, the results in respect to forest stand types and soil parent material groups especially for the organic layer indicated a relation between pH value and C stocks (Fig. 6.13). Both, soil parent material groups, representing a broad range of soil acidity, and forest stand types, representing a broad range of base and acidic cations caused by the litter of various tree species, showed a significant gradient of acidity. Moreover, C stock changes of the organic layer were negatively related to pH value. In forest soils liming is performed to compensate for acidic inputs and to improve the cation supply of the trees. Therefore, liming increases the pH value which in turn affects the organic carbon cycle in forest soils (see Sect. 6.5.3). Various studies demonstrated that nutrient-rich soils tend to be associated with higher rates of litter decomposition, whereas the accumulation of soil organic matter in less fertile soils occurs due to reduced decomposition (Oostra et al. 2006; Vesterdal et al. 2013). Vesterdal (1999) demonstrated that the accumulation of C in the organic layer was higher on less fertile sites than on fertile sites. This was explained by higher activity of organisms on fertile sites that incorporate material from the organic layer into the mineral soil. A study on Norway spruce stands on a soil poor in nutrient showed that the root mass in the forest floor was much greater
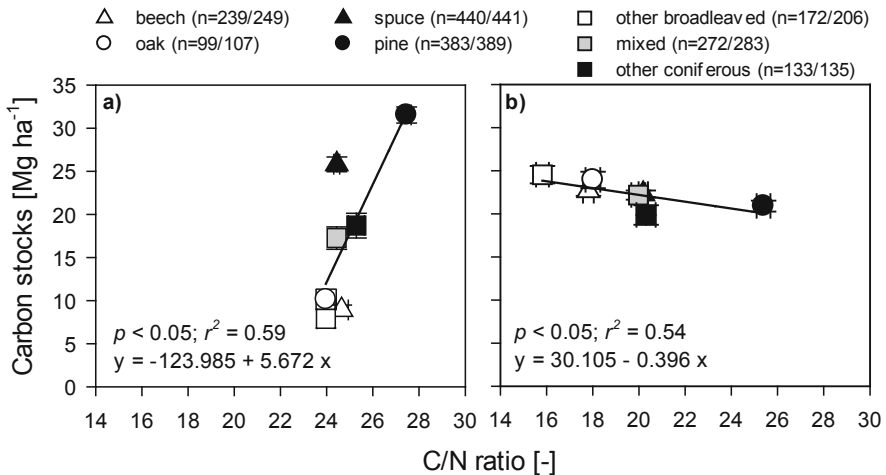
Fig. 6.14 Relation between the C/N ratio and carbon stocks (**a**) in the organic layer and (**b**) in the 0–5 cm depth increment of the mineral soil. The linear regression was performed between the averaged pH value carbon stocks of various forest stand types; the numbers in the legend represent forest stand types; for definition see Chap. 2

than in stands on more nutrient-rich soils (Vesterdal et al. 1995). A larger input of root litter to the organic layer at acidic sites poor in nutrient may thus additionally contribute to a higher sequestration of C in the organic layer.

6.5   Effects of Anthropogenic Factors

6.5.1   Forest Stand Structure

Significant differences in C stocks of the organic layer could be found for different tree species (Fig. 6.7a). Carbon stocks in the organic layer were highest under coniferous trees followed by beech and other broadleaf tree species. This order corresponds with findings compiled from various European and North American studies (Vesterdal et al. 2013). The results have in common that despite various methodological approaches, C stocks of the organic layers were affected by tree species. Differences in the accumulation of C can be explained by litter quality (Fig. 6.14). Various studies found higher C/N ratios in litter from coniferous trees compared to litter from broadleaf trees. Beech trees showed higher C/N ratios than litter from oaks and valuable broadleaved tree species (Vesterdal et al. 1995; Guckland et al. 2009). The C/N ratio of the litter can also serve as a proxy for other important chemical properties of the aboveground litter such as lignin content (Hobbie et al. 2006). The lignin/N ratio in turn is related to the turnover and accumulation of organic matter. The lignin/N ratios described in literature decreased

as followed: spruce, pine, beech > oak > other broadleaved trees (Kalbitz et al. 2006; Lovett et al. 2004).

The NFSI also confirmed an influence of tree species on C stock in mineral soil. We found a trend in sequestering more C in subsoils under broadleaf trees (30–90 cm). There are only a few studies on regional or national scale reporting tree species depending C accumulation in mineral soil. Some large-scale inventories included soil sampling and thus provide information on C stocks under different forest stands or tree species. Effects of tree species, however, might be masked by site-related factors (Vesterdal et al. 2013). Nevertheless, recent studies reported a tendency to store more C under broadleaf and valuable broadleaved tree species compared to coniferous trees and beech, respectively (Langenbruch et al. 2012; Oostra et al. 2006; Vesterdal et al. 2008). Tree species-related mechanisms and processes that affect C stocks could not be verified with the data from the NFSI. We suggest that the tree species could be driven by differences in soil fauna activity and related incorporation of litter into the soil. Variations in earthworm abundance between broadleaf and coniferous tree species have been reported by various studies (de Schrijver et al. 2012; Reich et al. 2005). These studies clearly reported that earthworm abundance or biomass increased from spruce over beech and valuable broadleaved tree species. We assume that earthworm abundance or biomass is a likely explanation for differences in C stocks between forest stand types or tree species due to higher inputs of organic layer material to upper mineral soil. A further mechanism explaining differences in mineral soil C stocks is deemed to be tree species-related differences in root biomass and its turnover. Temperate climate regions are known to have a similar C input through root biomass as aboveground C input through litter fall (Kleja et al. 2008) and to have significant higher below-ground biomass of broadleaf trees compared to coniferous trees (Finer et al. 2007). Furthermore, the C released by root litter and exudates is more stable than C that entered the soil by aboveground litter input (Crow et al. 2009).

The tree biomass was of minor importance explaining differences in C stocks and C stock changes of the organic layer (Fig. 6.10). Carbon stocks in the organic layer only slightly increased with tree biomass and accompanied increase of forest stand age. In contrast to our results, most of the studies could hardly reveal differences in C-related litter input of various tree species. Therefore in their studies, direct effects on organic layer C stocks could not be proved (Binkley and Valentine 1991; Trum et al. 2011; Vesterdal et al. 2008). Organic matter decomposition and turnover depend on the activity of decomposers which in turn is controlled by climate and/or forest management (Augusto et al. 2002). Higher tree biomass likewise is accompanied by a recovery of the site since the last forest management activity. A long-lasting period without disturbances due to forest management might thus favor accumulation of C (Schulze et al. 1999). Growing and yield charts suggest a decrease of forest stand productivity of aged stands resulting in steady-state equi-librium after decades. Studies on coniferous forests also showed an increase of organic layer mass with increasing forest stand age developing steady-state equilib-rium after several decades (Böttcher and Springob 2001). The age class distribution of forest stands illustrates that in our study the major part of forest stands are rather in

a growth period than in steady-state equilibrium. In contrast to the organic layer, we could not find a relation between tree biomass and C stocks for the mineral soil (Fig. 6.11). Investigations about effects of forest stand age, which is proportional related to coarse wood volume, revealed marginal correlations with C stocks (Böttcher and Springob 2001). Furthermore, based on yield charts, it can be derived a higher tree biomass of stands similar in age for coniferous than for broadleaf trees indicating that revealed effects might also be induced by differing tree species. This is supported, e.g., by a higher C accumulation of organic layers in coniferous forests compared to broadleaf forest. The negative relation between coarse wood volume and C stock changes in mineral soil is attributable to the fact that sites with high C stocks are established at soils from weathered carbonate rocks, loamy soils of the lowland, or soils of broad river valleys usually planted with broadleaf trees, whereas low C stocks can be found in soils from both base-poor unconsolidated sediment and bedrock usually planted with coniferous trees. Furthermore, soils under older forest stands may be rather in steady state of C input and output than soils under younger forest stands that have greater potential to fix C, and this difference may be attributed to a loss of organic matter following management disturbance (Covington 1981).

6.5.2   Atmospheric Nitrogen Deposition

A previous NFSI report discussed imbalances between pH value, C/N ratio, and humus forms that were attributed to atmospheric deposition of N (Wolff and Riek 1996). By implementing NFSI data into SEM, it could be shown that organic layer C stocks were positively correlated with N deposition, while organic layer C stock changes showed no response to N deposition (Fig. 6.11). More complex relations showed that both C concentrations and C stock as well as C stock changes in the mineral soil were affected by N depositions. This can be explained by the fact that the additional available N potentially increases the forest stand growth rates which in turn are likely to cause an accumulation of organic matter by enhanced biomass input into the soil. Furthermore, the oversupply of N stimulates microorganisms decomposing less C to receive sufficient nutrients. Long-term experiments in tempered and northern hardwood forest soils revealed a decrease in soil respiration under enhanced N deposition (Bowden et al. 2004; Burton et al. 2004). The positive relation between N deposition and C stocks and C stock changes revealed in the SEM might also be caused by collinearity with the data used for modelling N deposition especially for interactions of N deposition with height above sea level, geographical position, or forest stand type.
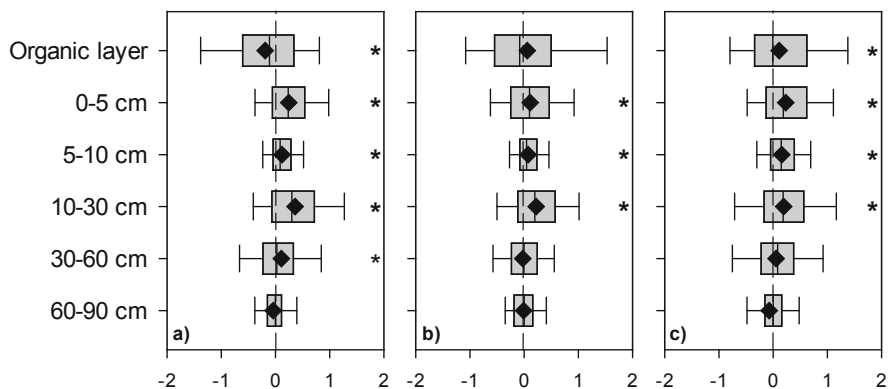
Fig. 6.15 Depth distribution of the annual organic carbon stock changes of (**a**) limed and (**b**) not limed sites sensitive to soil acidification as well as (**c**) sites both not limed and not sensitive to soil acidification. Asterisks indicate significant differences based on a one-sample *t*-test with H0: 0 ($p < 0.05$)

6.5.3   Forest Liming

The 1st NFSI report indicated a nationwide soil acidification widely independent from soil parent material (Wolff and Riek 1996). The consequences of soil acidification are manifold for eco-systematic functions of forests. Concepts for forest liming that consider specific site conditions were developed for an efficient and long-lasting regeneration of essential soil functions. The identified sites sensitive to soil acidification comprise sites that are suitable to lime according to specific liming concepts (see Chap. 2). Altogether, 749 out of 1859 survey plots of the NFSI were identified as sites sensitive to soil acidification including at least 385 survey plots limed since the 1st NFSI.

The results showed relations between liming and C stocks in the organic layer and mineral soil. We found in the organic layer at limed sites a significant decrease in C stock changes of 0.19 ± 0.05 Mg ha^{-1} year^{-1} (Fig. 6.15a). Increasing C stock changes of the organic layer were detected for not limed but sensitive sites to soil acidification with 0.06 ± 0.08 Mg ha^{-1} year^{-1} (Fig. 6.15b) and for not limed sites not sensitive to soil acidification with 0.11 ± 0.04 Mg ha^{-1} year^{-1} (Fig. 6.15c) sensitive to soil acidification. Taking the mineral soil into account, C stocks at limed sites increased significantly by 0.25 ± 0.04 Mg ha^{-1} year^{-1} in 0–5 cm, by 0.12 ± 0.02 Mg ha^{-1} year^{-1} in 5–10 cm, by 0.36 ± 0.05 Mg ha^{-1} year^{-1} in 10–30 cm, and by 0.11 ± 0.05 Mg ha^{-1} year^{-1} in 30–60 cm since the 1st NFSI. The gain of C in mineral soil thus offsets clearly the loss of C in organic layer. In respect to C stocks of the 1st NFSI, total soil organic C stocks throughout the whole soil profile showed a net increase of ~9%. A significant increase of 0.11 ± 0.05 Mg ha^{-1} year^{-1} in 0–5 cm, of 0.07 ± 0.02 Mg ha^{-1} in 5–10 cm, and of 0.22 ± 0.06 Mg ha^{-1} in 10–30 cm was also found at not limed forest sites sensitive to soil acidification. The increase, however, was less distinct as for the limed sites

Table 6.4 Results of analysis of covariance (ANCOVA) testing effects of clay content, temperature, and liming on organic carbon stock changes in the organic layer

Variables	DF	MSE (type III)	F value	p value
Clay content	1	18.800	17.02	<0.0001
Temperature	1	26.552	24.04	<0.0001
Liming	1	0.004	0.04	0.8433
Clay content*temperature	1	21.680	19.63	<0.0001
Clay content*liming	1	1.437	1.30	0.2544
Temperature*liming	1	0.239	0.22	0.6422
Clay content	1	18.800	17.02	<0.0001

DF degrees of freedom, *MSE* mean square error

and the C change of the soil profile since the 1st NFSI has totally increased by ~6%. Annual C stock changes of sites not sensitive to soil acidification showed significant increases in the organic layer of 0.11 ± 0.04 Mg ha^{-1} year^{-1} and in the mineral soil of 0.23 ± 0.03 Mg ha^{-1} year^{-1} in 0–5 cm, of 0.16 ± 0.02 Mg ha^{-1} year^{-1} in 5–10 cm, and of 0.20 ± 0.04 Mg ha^{-1} year^{-1} in 10–30 cm. The annual C stock changes were not significant in the subsoil (below 30 cm soil depth).

To exclude interactions with factors affecting C stock changes within the sites sensitive to soil acidification, multivariate analyses of covariance (ANCOVA) were performed for the organic layer as well as for the mineral soil (0–30 cm). We used for the organic layer the input variables liming, clay content (representing site properties), and temperature (representing climate). The ANCOVA revealed a clear influence of clay content and temperature on C stock changes in the organic layer (Table 6.4). The SEM in contrast has illustrated no temperature effects. On the other hand, the results of the ANCOVA showed an interaction between clay content and temperature. The differing results pointed out that C stock changes of the organic layer depended on soil parent material, although strength and direction of the relation were controlled by averaged mean temperature. No evidence was found for an influence of liming on C stock changes by multivariate analysis.

The SEM for mineral soil exposed effects of clay content and N deposition on C stock changes. This result was verified by ANCOVA in respect to effects of liming exclusively with the data from sites sensitive to soil acidification. The ANCOVA revealed direct effects of liming, while the influence of clay content and N deposition was not significant (Table 6.5). There were, however, significant interactions between clay content and liming as well as between clay content and N deposition. These findings indicated that liming favors the accumulation of C in mineral soils, strength, and direction, however, was depending from clay content and/or N deposition. The SEM supported the results of the ANCOVA revealing indirect positive as well as direct negative interdependences. The multivariate analysis also demonstrated that limed sites with low clay content accumulated high amounts of C, while limed sites with high clay concentrations tended to lose C. Similar results were found for N deposition where sites with high depositions sequestered C, while sites with low depositions were affected by a loss of C. This suggests that limed sites that are characterized by high N deposition are different in turnover and

Table 6.5 Results of analysis of covariance (ANCOVA) testing effects of clay content, temperature, and liming on organic carbon stock changes in the mineral soil down to a depth of 30 cm

Variables	DF	MSE (type III)	F value	p value
Clay content	1	6.213	2.77	0.0966
N deposition	1	0.107	0.05	0.8273
Liming	1	16.030	7.14	0.0077
Clay content*N deposition	1	5.887	2.62	0.1058
Clay content*liming	1	17.091	7.61	0.0059
N deposition*liming	1	11.839	5.27	0.0219

DF degrees of freedom, *MSE* mean square error

decomposing soil organic matter as sites that are either exclusively limed or affected by high N deposition. Especially not limed sites close to arable land or to road traffic might be affected by higher N inputs.

The calcium applied by the lime plays an important role in forest ecosystems and therefore can affect directly C pools and fluxes. The loss of C of limed organic layers compared to organic layers not limed may be caused by changes in the microbial and faunal composition and function. It is known that the microbial activity (Illmer and Schinner 1991; Andersson and Nilsson 2001) and the earthworm abundance (Hobbie et al. 2006; Reich et al. 2005) are stimulated when the pH value and calcium concentrations increase. The increasing C sequestration rate of the mineral soil on limed sites can be attributed to an enhanced calcium availability through the application of lime, which may reduce the dissolution and mobility of dissolved organic matter by forming cation bridges (Oste et al. 2002). These cation bridges are thought to stabilize soil organic matter and reduce decomposition which in turn causes enhanced retention of soil organic matter. Studies about effects of liming on soil respiration have observed a decline in respiration after an initial enhancement period to a few years post-liming (Illmer and Schinner 1991; Melvin et al. 2013; Groffman et al. 2006). On the other hand, observations of increased decomposition with liming have been reported in various studies (Baath and Arnebrant 1994; Priha and Smolander 1994). Nevertheless, reasons for these differential responses remain unclear. A reduction of soil respiration indicates that liming is affecting relations between microbial communities and the organic matter mineralized. Changes in microbial communities by liming may cause changes of recalcitrance of produced soil organic matter or physical stabilization of soil organic matter.

6.6   Summary and Conclusions

The analysis of the data from NFSI revealed a C sequestration rate of 0.75 Mg ha^{-1} in the organic layer and the mineral soil down to 90 cm for the time period between the inventories. Especially the organic layer and depth increments down to 30 cm depth showed an increase, while in the subsoil (below 30 cm soil depth), changes were marginal. With respect to climate protection policy, the sequestration of atmospheric

C as stable long-lasting soil organic matter is of interest due to the fact that our subsoils alone stored 25% of total C stocks. Carbon stocks of the organic layer and the mineral soil down to 90 cm totaled to 117.1 Mg ha^{-1} in 2006, of which 17% of organic C was stored in the organic layer and 58% in the upper 30 cm of the mineral soil.

The results showed that the variation of C stocks in the organic layer was especially controlled by tree species as soils under broadleaf trees stored less C in the organic layer than soils under coniferous trees. In contrast, tree species effects on C stocks of mineral soil were comparatively less pronounced. This may be explained by both differences in litter acidity and nutrient availability. Sites with broadleaf trees may store more C in subsoil compared to sites with pine and spruce trees and are therefore predestined for long-term C sequestration. This is probably due to the fact that there is under broadleaf and mixed forests a higher input of fairly stable organic matter to deeper soil horizon by deeper rooting trees, a higher biological activity, and/or less disturbance of the soil caused by more stable stands. Soil C stocks are furthermore affected by site conditions and the resulting soil texture. An effective selection of tree species combined with specific site conditions may therefore enhance C sequestration of mineral soils. This can probably be facilitated by establishing conifer mixed forests or by specific placements of broadleaf tree species in pure coniferous stands especially on soils derived from unconsolidated sediments and/or on soils derived from base-poor bedrock. Besides the analysis of C input and output (mineralization, leaching, etc.), the quantitative estimation of soil organic matter stabilization has to be investigated in more detail to understand processes of C sequestration in forest soils.

In explaining more or less large-scaled patterns of changes in soil organic C stocks, various regional factors have to be considered. Wide areas of the Northern German Lowland are characterized by weakly developed sandy soils derived from pre-Pleistocene sediments that were depleted of soil organic matter by extensive overuse. The site quality, however, has improved by reversing the degradation of conscious forest management since NFSI I. In particular cases it can be assumed that rather locally effective processes are crucial in explaining C stock changes. The increase or decrease of C stocks in the organic layer and mineral soil can be controlled by forest management such as forest liming, tree species selection, and forest conservation. Besides natural environmental effects of soil parent material, texture, and increasing climate-induced water stresses, the pathway analysis revealed additional specific effects of N deposition on C stock changes. Furthermore important is the canopy uptake or the proximity to arable land in respect to the total deposition. Nevertheless, both factors could not be integrated in the model and have to be disregarded. In general, terrestrial ecosystems respond to enhanced N deposition when they are N limited. The additional N has the potential to increase sequestering C by an increase in productivity and accumulation of soil organic matter through increased litter production. Carbon accumulation through N deposition may also be due to increased recalcitrance of N-enriched litter and decreased soil respiration, both leading to reduced long-term decomposition rates of organic matter. Furthermore, modelling N input is affected by covariance, e.g., between climate and N deposition, because the amount of N deposed at a site depends on in situ

precipitation. The relationship between climate variables and N deposition could be further assessed by simultaneously measuring N stocks and fluxes when aspects of climate and deposition may be recorded separately as conducted by the intensive (Level II) monitoring.

Besides timber harvest and decommissioned forests, a further aspect discussed in forest ecology is the application of lime to reduce negative consequences of atmospheric deposition of acids. Our results showed that liming may affect C cycling in forest ecosystems as C stocks increased in mineral soil, while a loss of C was observed in the organic layer. In literature various results have been discussed in light of the impact of liming on soil organic C cycling in forest ecosystems. Recent research showed that soil respiration was both stimulated and inhibited by forest liming depending on various environmental conditions. Further research is needed to identify the most important factors affecting turnover of soil organic matter in respect to the impact of forest liming especially on dynamics of microbial communities as well as on recalcitrance and stabilization of soil organic matter. A prolonged impact on C sequestration and N retention is expected considering Germany received a long-term and area-wide, largely substrate-independent acidification and soil depletion through atmospheric inputs of acids. From this point of view, more detailed research is needed studying effects of the application of cations by liming on C pools and fluxes in forest ecosystems.

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