



Assessment of carbon-biodiversity change from forestry projects in Republic of Korea

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Background: This study was undertaken to analyze the impact of reforestation and logging (i.e., forestry projects) on forest carbon stocks and biodiversity. Vegetation and biodiversity data were utilized from areas where forestry projects were implemented to estimate changes in carbon stocks and biodiversity before and after forestry projects. The carbon-biodiversity linkage assessment was developed by indexing carbon stocks and biodiversity in the same ratio.

Results: A high carbon-biodiversity linkage assessment index indicates high carbon stocks and biodiversity. Forestry projects were found to have a negative impact on both carbon stocks and biodiversity. Significant reductions in tree above-ground and soil carbon stocks and biodiversity declines of most taxonomic groups were observed. However, due to differences in ecosystem characteristics, the magnitude of biodiversity decline varied among taxa. A decline following forestry projects was demonstrated as a result of the carbon-biodiversity linkage assessment. A decrease in the carbon-biodiversity linkage index indicates a weakening of ecosystem services, climate change reduction, and mitigation functions of the target area. Through carbon-biodiversity linkage assessment, this study identified hotspot areas with high carbon stocks and biodiversity.

Conclusions: These areas of concern can provide a policy basis information for simultaneously increasing and conserving carbon storage capacity and biodiversity in Korea. Therefore, we recommend an ecosystem survey database that is consistent in time and space should be established and managed for accurate carbon-biodiversity linkage assessment. It is also necessary, depending on ecosystem type and environmental impacts, to investigate the synergy and trade-offs between carbon stocks and biodiversity.

Keywords: carbon storage, forest management, species diversity, synergy, tradeoff

Introduction

Globally, biodiversity trends continue to decline and, in ways that threaten human survival, anthropogenic greenhouse gas (GHG) emissions are changing the Earth's climate (Finn et al. 2023). Climate change is a cause of biodiversity decline, and its intensity is increasing (Alshehab 2024). In contrast, biodiversity and ecosystem functioning can contribute significantly to adaptation and mitigation of climate change (Gomasasca et al. 2023). Therefore, through strengthening biodiversity and ecosystem functioning, the international community has been working to restore ecosystems and mitigate climate change. The 15th Conference of the Parties (COP15) to the Convention on Biological Diversity (CBD) adopted an action target to manage 30% of terrestrial and marine ecosystems as protected areas and restore degraded ecosystems for biodiver-

sity conservation by 2030. Furthermore, the Paris agreement, signed at the 2015 United Nations Framework convention on climate change (UNFCCC), aims to reduce GHG emissions and strive to limit the temperature increase to well below 2 degrees Celsius above pre-industrial levels. In particular, the UNFCCC seeks to limit the increase to a possible 1.5 degrees Celsius (IPCC 2018). Republic of Korea has established the 1st National Framework Plan for Carbon Neutrality and Green Growth (2023–2042) to realize a carbon-neutral society. This framework proposes policies to reduce GHG emissions and strengthen the foundations for implementation. Among these foundations, forests have been selected as a key project, to fulfill GHG reduction targets, by improving the quantity and quality of the sink and systematic management of sink restoration to enhance their function.

However, despite national and international efforts to



protect ecosystems and mitigate climate change, development projects continue to disturb and damage Korean ecosystems. Specifically, forest cover in Korea comprises approximately 64% of the country's total land area (NIFOS; National Institute of Forest Science 2022), making forest ecosystems a major carbon sink and habitat for biodiversity. The number of development projects affecting ecosystems in Korea over the last 10 years (2014–2023) was about 1,268, of which about 264 (20.8%) were mainly carried out in mountainous areas. These projects included works such as mountain development, extraction of soil, sand, and gravel minerals, and energy development (EIASS; Environmental Impact Assessment Support System, www.eiass.go.kr 2023). Moreover, the Korean Forest Service has established a plan to improve the age structure of trees and open forest roads to strengthen carbon absorption capacity by 2050 through the harvesting of old trees with low carbon absorption capacity and planting of young trees with high carbon absorption capacity (Korea Forest Service Press Release 2021.1.20). This implementation of this plan is expected to further increase the disturbance of Korean forests in the future. The intensive reforestation efforts in the 1970s and 1980s led to an unbalanced age-class distribution in forests, necessitating strategic logging and reforestation plans to ensure sustainable timber production and ecological benefits. These activities can lead to changes in forest structure, including impacts on carbon storage and water conservation capacities (Vásquez-Grandón et al. 2018). Additionally, the construction of forest roads disrupts soil and vegetation, increasing erosion, decreasing biodiversity, and altering natural forest regeneration (Lee and Yun 2024).

Due to degradation, physical changes in forests reduce forest biodiversity and carbon storage capacity. Thus, accurate assessments of the extent of change are important. Previous research studies that have assessed the linkages between ecosystem biodiversity and carbon storage include high-resolution mapping of the co-benefits of carbon stocks and biodiversity values in vegetation and soils worldwide, as well as using terrestrial biodiversity and carbon storage datasets to map potential synergies between carbon and biodiversity (Soto-Navarro et al. 2020; Strassburg et al. 2010). O'Brien et al. (2023) analyzed Canadian hotspot areas with high levels of both carbon storage and landscape connectivity to protect biodiversity from climate change. Vijay et al. (2022) studied the importance of policy decisions and examined synergy and tradeoff relationships using data on habitat area, species richness, endangered species richness, and carbon stocks. However, there has been no related research on carbon-biodiversity assessments in Korea.

The purpose of this study is to assess the linkage between carbon storage and biodiversity changes before and after the implementation of forestry projects. Specifically,

it aims to verify changes in vegetation and biodiversity both spatially and temporally within areas affected by forestry activities, to evaluate the correlation between changes in carbon storage and biodiversity. This analysis helps to scientifically understand the impact of forest management activities on ecosystems and contributes to developing effective forest management and conservation strategies.

Materials and Methods

Selection of study site

This study provides a linkage assessment of carbon storage and biodiversity changes before and after forestry project implementation. Thus, it is necessary to obtain data that confirms changes in vegetation and biodiversity, both spatial and temporal, due to forestry projects. First, to select the study sites, forest maps were collected with basic information (area, year of implementation, type of project) of forest projects (i.e., reforestation, logging, thinning). The Korea Forest Service produced a 1:25,000 scale forest map that depicts forest distribution in Korea and includes attribute information such as forest type, tree species, Diameter at Breast Height (DBH) class, age class, and tree density. Furthermore, the Korea Forest Service is compiling a spatial database for each type of forest management activity and is collecting 1.12 million forest management activity database entries by 2023. This information is demonstrated in the forest type map. Based on the forest type map, this study extracted and utilized the forest management activity data accumulated until 2022. Our study identified and collected data from various biodiversity surveys conducted in Korea to select areas where biodiversity survey data can be compared before and after forestry project implementation. Specifically, we utilized the results of the National Ecosystem Survey (NES), a countrywide survey of eight categories of plants, mammals, birds, insects, reptiles, amphibians and fishes, occurring every five years. These results of the NES were divided into pre-project (1997–2005) and post-project (2014–2018) based on the survey period. Based on the survey period, we selected five regions with well-defined forestry and reforestation information and relatively intact biological taxonomic group information to evaluate carbon and biodiversity changes due to forestry projects.

Description of study site

The selected sites were at Daeryang (Site 1; 35°52'–35°49'N, 127°32'–127°34'E, 183.5 ha), Andeok (Site 2; 36°19'–39°17'N, 128°55'–128°57'E, 124.2 ha), Sinjeong (Site 3; 35°49'–35°47'N, 127°19'–127°22'E, 116.6 ha), Mugye (Site 4; 35°12'–35°10'N, 128°55'–128°57'E, 78.7 ha), and Gwanghyewon (Site 5; 36°57'–36°55'N, 127°22'–127°25'E, 77.5 ha). In addition, two sites in Jinan city and one each in

Cheongsong city, Jincheon city, and Gimhae city were chosen (Fig. 1). Table 1 demonstrates the forest status of each site. The total forest area of Site 1 was 183.5 ha, and the vegetation before the forest project was composed of *Quercus acutissima*, *Pinus densiflora*, *Zelkova serrata*, *Chamaecyparis pisifera* and *Liriodendron tulipifera*. The highest age class area, age class IV (31–40 years old), accounted for 15.4% of the total area. Post-forestation vegetation was dominated by *Q. acutissima* and *Fraxinus rhynchophylla*, with 0.6% of the area in class IV or higher. The total forest area of Site 2 was 124.2 ha, and the pre-forestry vegetation comprised of *Quercus*, *P. densiflora*, and *L. tulipifera*, with 3.6% of the total area in class IV or higher. Post-forestation vegetation was characterized by *Quercus*, *P. densiflora*, and *L. tulipifera*, with no areas of class 4 or higher. Total forest area of Site 3 was 116.6 ha, and pre-forestry vegetation was composed of *Q. acutissima*, *P. densiflora*, and *Chamaecy-*

paris obtusa. The area of class IV and above was 22.9% of the total area. After the forestry project was completed, the vegetation was mainly *Q. acutissima*, with 1.5% of the area in class IV and above. The total area of Site 4 forest was 78.7 ha, and pre-forestry vegetation was *Prunus serrulata* and *C. obtusa*, etc. The area of class 4 and above was 62.9% of the total area. After the forest project completion, the vegetation was composed of *P. serrulata* and *C. obtusa*, etc. and the total area of class IV and above increased to 100%. The total forest area of Site 5 was 77.5 ha, and the vegetation prior to the forest project was *Pinus koraiensis* and *C. obtusa*, etc. The area of class IV and above was 18.7% of the total area. After the forest project was completed, *P. densiflora*, *P. koraiensis*, and *C. obtusa* comprised the vegetation, and there were no observed areas of class IV or higher. Additionally, the age class was grouped at 10-year interval (i.e., I: 1–10 years old, II: 11–20 years old, III: 21–30 years old, IV: 31–40 years old, V: 41–50 years old, VI: 51–60 years old).

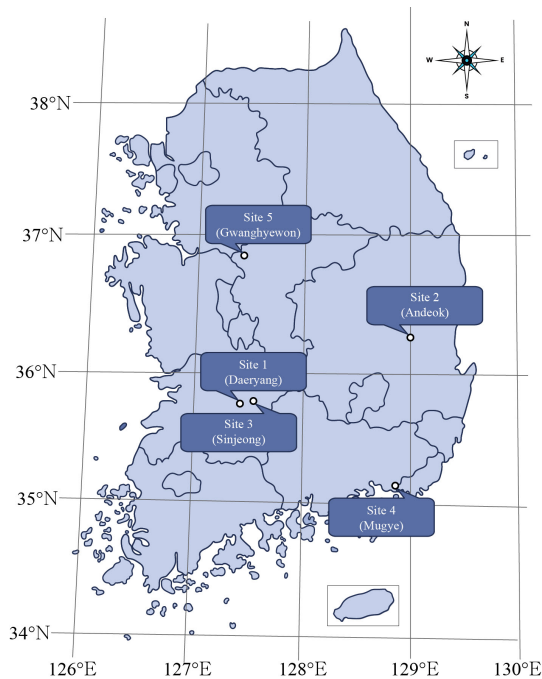


Fig. 1 Locations of study sites.

Biodiversity index

Biodiversity assessments data

For the biodiversity assessment, this study utilized species data established by the NES. The NES is a survey conducted in accordance with the Natural Environment Conservation Act to systematically conserve and manage Korea's natural ecosystem (National Institute of Ecology 2020). Since the NES began in 1986, it has been conducted five times through 2023. The 6th National Ecosystem Survey (2024–2028) is currently underway. Altogether, the NES surveys nine biological taxonomic groups (vegetation, flora, birds, mammals, amphibians, reptiles, terrestrial insects, benthic macroinvertebrates, and freshwater fishes) across the country. Chronologically, the NES has accumulated about 37 years of data in the five surveys since its inception (1986–1990, 1997–2005, 2006–2013, 2014–2018, and 2019–2023). The NES is conducted using the same survey methodology by taxonomic group and order in inland, river, and coastal areas. The biodiversity data in this

Table 1 Status of dominant vegetation in study regions before and after of forestry project implementation

Site no.	Area (ha)	Before forestry project		After forestry project	
		Dominant vegetation	Ratio of IV–VI ^a area (%)	Dominant vegetation	Ratio of IV–VI area (%)
1	183.5	<i>Quercus acutissima</i> , <i>Pinus densiflora</i> , <i>Zelkova serrata</i> , <i>Chamaecyparis pisifera</i> , <i>Liriodendron tulipifera</i>	15.4	<i>Q. acutissima</i> , <i>Fraxinus rhynchophylla</i>	0.6
2	124.2	<i>Quercus</i> , <i>P. densiflora</i> , <i>L. tulipifera</i>	3.6	<i>Quercus</i> , <i>P. densiflora</i> , <i>L. tulipifera</i>	0
3	116.6	<i>Q. acutissima</i> , <i>P. densiflora</i> , <i>C. obtusa</i>	22.9	<i>Q. acutissima</i>	1.5
4	78.7	<i>Prunus serrulata</i> , <i>C. obtusa</i>	62.9	<i>P. serrulata</i> , <i>C. obtusa</i>	100
5	77.5	<i>Pinus koraiensis</i> , <i>C. obtusa</i>	18.7	<i>P. densiflora</i> , <i>P. koraiensis</i> , <i>C. obtusa</i>	0

^aIV means forest that ratio of crown as 31–40 age trees is more than or equal to 50%. V means forest that ratio of crown as 41–50 age trees is more than or equal to 50%. VI means forest that ratio of crown as 51–60 age trees is more than or equal to 50%.

study represented seven taxonomic groups: birds, mammals, amphibians, reptiles, terrestrial insects, benthic macroinvertebrates, and freshwater fishes. The survey data of NES can be downloaded from the Ecobank website (nie-ecobank.kr) of National Institute of Ecology.

Biodiversity assessment data are collected in different forms for different purposes and are recorded in different ways. To be able to effectively use these various survey data entry methods for biodiversity assessment, survey data was pre-processed into a single form. Pre-processing included: correcting spelling errors of Korean and scientific names as well as similar Korean and scientific names previously used by referring to the National Species List published by the National Institute of Biological Resources, removal of symbols such as degrees, minutes, and seconds in geographic coordinate system from the input data, and converting degrees, minutes, and seconds (DMS) coordinates to longitude and latitude decimal degree coordinates, and converting various survey dates of the survey year to yyyy-mm-dd format, and extracting only the survey year (yyyy) afterward. After this process, the data was built into a database with the Excel program (Microsoft, Redmond, WA, USA).

After pre-processing, the species location data were expressed in longitude-latitude coordinates (WGS84, EPSG: 4326). These coordinates express location on the earth in the form of a sphere. To represent the coordinates on a two-dimensional plane to build integrated species survey data by taxonomic group, they are then converted to the standard planar projection coordinate system of Korea's National Geographic Information Institute (Korea 2000 Central belt 2010, EPSG: 5186).

Biodiversity assessment methods

Biodiversity Assessment: The assessment was conducted before the forestry project was implemented (before 2006) and after (after 2014) to understand the impact of the forestry project on biodiversity. The forestry project period ran from 2006 to 2013, with tree felling and reforestation taking place in each site. The number of species present in each of the seven taxonomic groups (mammals, birds, reptiles, amphibians, freshwater fish, invertebrates, and insects) occurring within the study area was calculated by the biodiversity assessment. Then, to normalize the differences in total species size and species number by taxonomic group, the ratio was calculated between the number of species found within each site and the total number of species by taxonomic group found at the time of the survey and then used as a biodiversity value. The calculation formula is shown in Equation (1).

$$S' = \frac{S_{site}}{S_{total}} \times 100 \quad (1)$$

※ S_{total} = Total number of individual taxa surveyed during a particular survey period

S_{site} = Same survey period as S_{total} , but number of individual taxa surveyed per assessment unit

Forestry project impacts on biodiversity were measured by comparing the change in species richness prior to the forestry project commencement to the difference in species richness after completion of the forestry project.

Carbon storage assessment

Above-ground carbon storage

In this study, we adapted aboveground carbon stocks for the carbon-biodiversity linkage assessment from the methodology and results of Jeong et al. (2023). The Jeong et al. (2023) study covers the same sites and time period as this study.

To estimate aboveground carbon stocks for Sites 1–5, we used tree volume (V), biomass expansion factor (BEF), wood density (WD), and carbon fraction (CaF). These factors are estimated using the equation (2) below. This method, introduced in the IPCC's Guidelines for GHG Inventory, is one of the methods used to estimate the biomass storage of forest trees (IPCC 2006).

$$\text{Carbon Stock} = V \times \text{BEF} \times \text{WD} \times \text{CaF} \quad (2)$$

※ V = Volume ($\text{m}^3 \text{ha}^{-1}$)

BEF = Biomass expansion factor (Needle-leaved tree: 1.43, Broad-leaved tree: 1.51)

WD = Wood density (ton ha^{-1})

CaF = Carbon fraction (Needle-leaved tree: 0.51, Broad-leaved tree: 0.48)

Predominantly, we use the actual tree volume data collected before and after logging at the target project site. However, this study utilized the growth coefficient ($\text{m}^3 \text{ha}^{-1}$) since it is difficult to obtain actual data for large areas of forests. The growth coefficient ($\text{m}^3 \text{ha}^{-1}$) quantifies the relationship between the characteristics of the stand (forest type, mature, tree trunk diameter, and dense) and the volume (Kim et al. 2014). This growth coefficient is determined by the corresponding volume value ($\text{m}^3 \text{ha}^{-1}$) based on the type of stand (broadleaf, coniferous, mixed forest), age class (I–VI), DBH class (small, medium, large), and density (low, medium, high). The forest data as type of stand, age class and density, can be downloaded at from Korea Forest Service website (www.forest.go.kr). Therefore, it is possible to determine the tree volume value of the study area indirectly through the stand information even if there is no actual data on tree volume. In this study, forest carbon storage was calculated according to the above equation (2) by utilizing the growth coefficient corresponding

to the specific forest information.

According to the forest sector GHG inventory calculation method, BEF was calculated by applying 1.43 for conifers and 1.51 for broadleaf (Ministry of Environment 2023). Similarly, WD was also calculated by applying 0.46 for conifers and 0.68 for broadleaf according to the forest sector GHG inventory methodology. The change in carbon stocks before and forest projects was estimated as the difference between the historical forest inventory information (before the forest project, 2005) and the post-forest project information (2022).

Soil carbon storage

The methodology and results of Jeong et al. (2023) were utilized to partially adapt soil carbon stocks for the carbon-biodiversity linkage assessment.

The basic calculation of the difference in soil carbon storage is the difference in carbon storage prior to and following completion of the forestry project. The amount of carbon storage in the soil is determined by soil depth (T), bulk density (BD), organic carbon content (C), and coarse fragments (CoF). Equation (3) shows the method for calculating soil carbon storage below.

$$CS(\text{Mg ha}^{-1}) = T \times BD \times C \times (1 - \text{CoF}) \quad (3)$$

※ T = Soil depth (cm)

BD = Bulk density (g cm^{-3})

C = Carbon content

CoF = Coarse fragment (%)

To analyze soil carbon stocks in forestry projects, all of the aforementioned data should be available for all sites as, to derive accurate values, it is necessary to use actual data from the target area. However, since it is difficult to obtain actual data for most logging projects, we calculated the values using data published by relevant national organizations or used in previous studies. We used the Forest Soil Map (<http://data.nsd.gov.kr/dataset/20180918ds00066>) published by the Korea Forest Service for soil thickness and stone content. For the organic carbon content in the soil, we obtained relevant information from the soil map (<https://soil.rda.go.kr> – site name is Heulgtoram) provided by the Rural Development Administration. However, since Heulgtoram (<https://soil.rda.go.kr>) only provides values for soil organic content, it was necessary to convert it to the amount of organic carbon by dividing it by 1.724. It is not possible to directly obtain soil BD from published data, so it must be obtained from parent rock information. Parent rock information can be obtained from the forest stand soil map, and soil BD is inferred from the parent rock using the information presented in Jeong et al. (2003) (Table 2). Soil depth was obtained from the Forest soil map.

As mentioned previously, pre- and post-forestry data are

Table 2 Bulk density and soil depth using Forest Soil Map

Site no.	Bulk density (g cm^{-3})	Soil depth (cm)
1	0.84	19.72
2	0.97	12.13
3	0.97	20.76
4	0.97	13.17
5	0.91	13.57
6	0.84	20.54

needed to analyze changes in soil carbon stocks due to forestry projects. However, the publicly available data described above are limited in scope, and there are acquisition limitations in pre- and post-forestry data. Thus, the soil carbon values estimated from the published data were assumed to be the pre-forestry values, and the post-forestry values were calculated under the assumption that they decrease by a constant percentage (Nave et al. 2010). A meta-analysis was conducted by Nave et al. (2010) of more than 6,500 studies of logging, harvesting, and clearcutting in temperate forests obtained through searches of online databases (ISI Web of Science, BIOSIS, etc.). Soil carbon reductions of 20% in coniferous and mixed forests and 36% in broadleaf forests at 5 to 20 cm soil depth were reported by the Nave et al. (2010) analysis. In our study, these percentages were used to estimate the reduction of soil carbon from forestry projects.

Carbon-biodiversity linkage assessment

The final goal of the study was to assess and index how both carbon and biodiversity are affected before and after forestry projects are undertaken. However, the index was calculated by considering carbon and biodiversity with equal weight since they are not directly related. Since seven taxonomic groups were targeted, biodiversity would have a distribution of -700 to 700%, and carbon would have a distribution of -200 to 200%. Therefore, each was divided by 7 and 2 to reflect the same proportion. The final sum was expressed as a score out of 10. Consequently, the linkage assessment result ranges from -10 to 10. A positive value indicates good results in terms of carbon storage and biodiversity, while a negative value indicates more of an adverse effect on biodiversity and carbon storage than before the linkage assessment.

Statistical analysis

This study utilized paired t-tests to analyze differences in forest carbon stocks due to forest projects. A paired t-test is a statistical method of analysis that examines the difference between two populations that correspond to each other one-to-one. Our study aimed to analyze the statistical significance of the change in carbon stocks due to forest projects (Ross and Willson 2017). Paired t-test analyses were performed using the R program (ver. 4.3.1, www.r-project.org) at a 5% significance level.

Results

Biodiversity changes due to forestry projects

When comparing data from before and after the forestry project, most taxa showed no or decreased biodiversity. This indicated a negative impact on biodiversity due to the forest projects (Table 3). At Site 1, following the forestry project, a biodiversity decrease was observed for amphibians (-22.2), mammals (-17.0), and freshwater fish (-5.6). At Site 2, the decreased biodiversity was greatest for reptiles (-26.6), followed by amphibians (-17.3). At Site 3, biodiversity was observed to decrease for mammals (-15.9) and freshwater fish (-3.9). At Site 4, biodiversity decreases were exhibited in amphibians (-33.3), freshwater fish (-14.6), and reptiles (-13.6). At Site 5, biodiversity decreased in the following order: amphibians (-38.9), reptiles (-31.8), and mammals (-22.6). As shown in Figure 2, the largest biodiversity decline was in amphibians (-17.3 to -38.9), followed by reptiles (-13.6 to -31.8) and mammals (0 to 22.6) across all sites. The taxa with the smallest observed declines were

terrestrial insects (-1.4 to 5.2), birds (-0.8 to -1.7) and benthic invertebrates (-4.2). Contrary to the other sites, terrestrial insects at site 2 experienced an increase in biodiversity after the forest project was completed.

Changes in carbon stock due to forestry projects

Before the initiation of forestry projects, above-ground carbon stock calculations showed that Sites 1, 2, 3, 4, and 5 were 48.7, 52.6, 55.2, 68.0, and 59.7 Mg C ha⁻¹, respectively (Fig. 3). Moreover, following completion of the forestry projects, above-ground carbon stocks were 18.0, 22.1, 10.1, 9.6, and 10.0 Mg C ha⁻¹, respectively. All study plots revealed a decrease in above-ground carbon stocks after forest project implementation ($p < 0.001$). Above-ground carbon stocks (percent reduction) at Sites 1–5 were 30.7 (63.1%), 30.5 (58.0%), 45.1 (81.7%), 58.4 (85.8%), and 49.6 (83.2%) Mg C ha⁻¹, respectively.

Soil carbon stock calculations showed that pre-forestry soil carbon stocks for Sites 1, 2, 3, 4, and 5 were 28.0, 48.1, 17.3, 21.5, and 27.2 Mg C ha⁻¹, respectively (Fig. 4). Decreases

Table 3 Comparison to biodiversity before and after forestry project

Site no.	Mammal			Bird			Amphibian			Reptile		
	Before	After	ΔS	Before	After	ΔS	Before	After	ΔS	Before	After	ΔS
1	17	0	-17	1.2	0	-1.2	22.2	0	-22.2	-	-	-
2	22.6	20.8	-1.8	-	-	-	27.8	10.5	-17.3	40.9	14.3	-26.6
3	18.9	3.8	-15.1	-	-	-	-	-	-	-	-	-
4	1.9	1.9	0	1.2	0	-1.2	33.3	0	-33.3	13.6	0	-13.6
5	22.6	0	-22.6	0.8	0	-0.8	38.9	0	-38.9	31.8	0	-31.8

Site no.	Freshwater fish			Macroinvertebrate			Terrestrial insect		
	Before	After	ΔS	Before	After	ΔS	Before	After	ΔS
1	5.1	0	-5.1	4.2	0	-4.2	1.2	0	-1.2
2	7.6	7.4	-0.2	-	-	-	2.6	7.8	5.2
3	7	3.1	-3.9	4.2	0	-4.2	1.4	0	-1.4
4	14.6	0	-14.6	4.2	0	-4.2	1.2	0	-1.2
5	1.9	0	-1.9	-	-	-	0.5	0	-0.5

A negative ΔS value indicates decline of biodiversity after the forestry projects, while a positive value indicates an increment of biodiversity after a forestry projects.

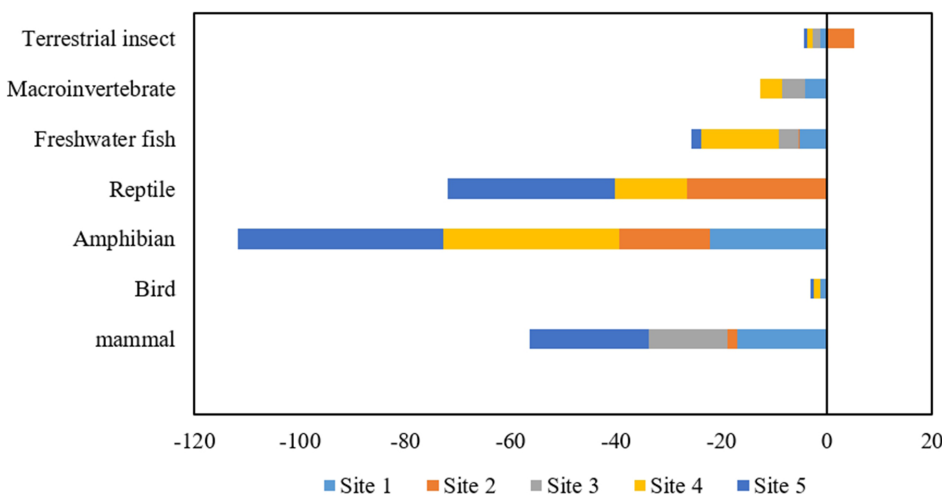


Fig. 2 Cumulative change in biodiversity by forestry projects.

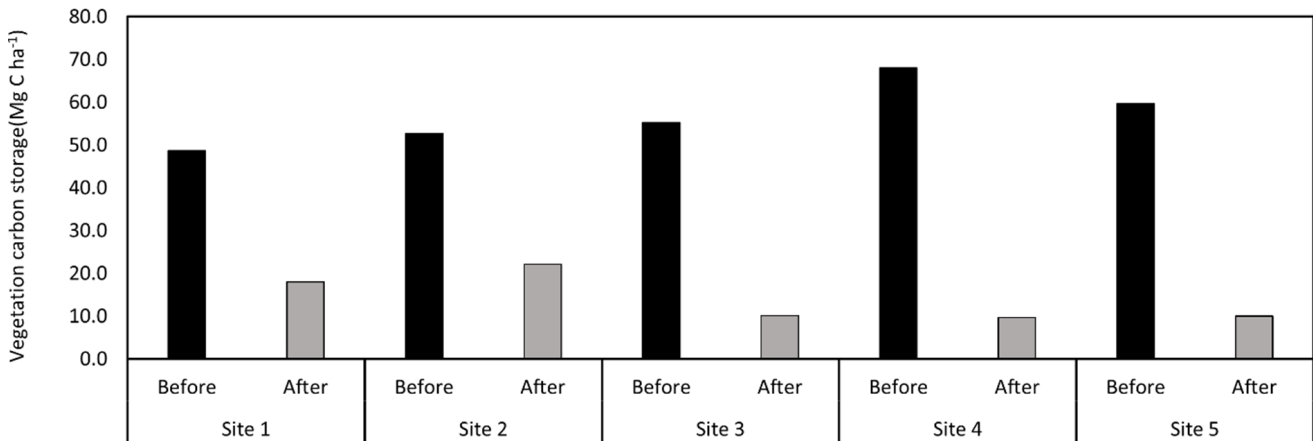


Fig. 3 Carbon stock (Mg C ha⁻¹) of above-ground before and after forestry project. The carbon stock decreased after the forestry project compared to before ($p < 0.0007$).

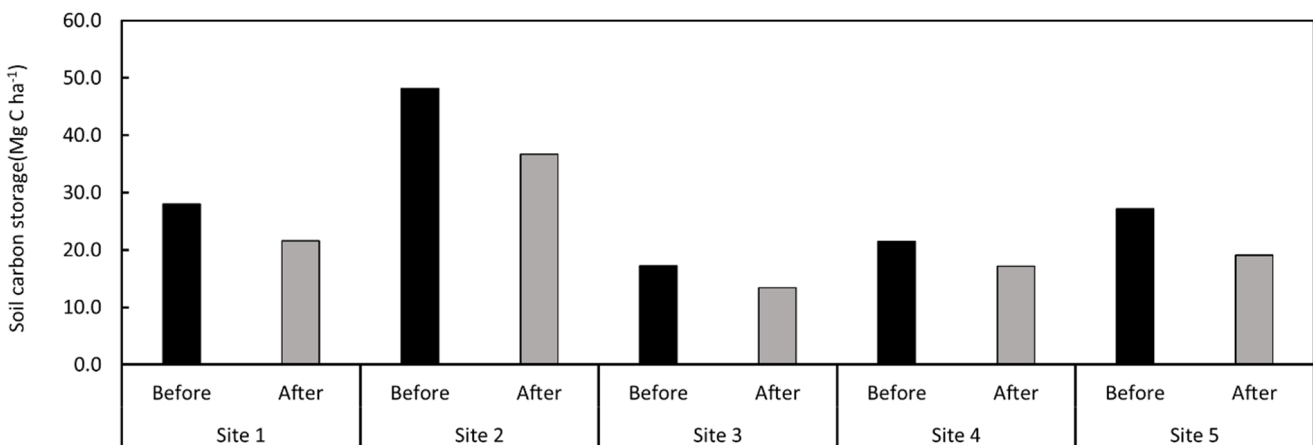


Fig. 4 Carbon stock (Mg C ha⁻¹) of soil before and after forestry project. The carbon stock decreased after the forestry project compared to before ($p < 0.002$).

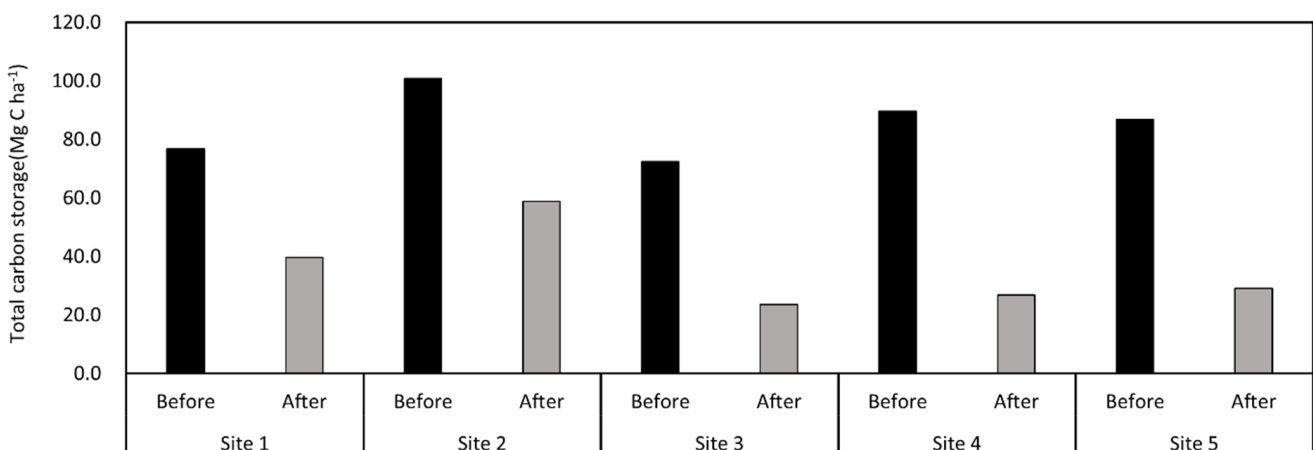


Fig. 5 Total carbon stock (Mg C ha⁻¹) before and after forestry project. The carbon stock decreased after the forestry project compared to before ($p < 0.0003$).

es in soil carbon stocks were revealed after the forest projects were completed in Sites 1–5, with results of 21.6, 36.6, 13.4, 17.2, and 19.1 Mg C ha⁻¹, respectively ($p < 0.01$). For Sites 1–5, the decrease in soil carbon (percent reduction) was 6.4 (22.9%), 11.4 (23.8%), 3.9 (22.3%), 4.3 (20.0%), and

8.1 (29.8%) Mg C ha⁻¹, respectively.

The pre-forestation above-ground and soil carbon stocks for Sites 1–6 were 76.7, 100.7, 72.4, 89.5, and 86.8 Mg C ha⁻¹ (Fig. 5), respectively. After the forest project was completed, total forest carbon stocks were calculated to be 39.6,

58.7, 23.5, 26.8, and 29.1 Mg C ha⁻¹, respectively. The total forest carbon stocks decreased in all assessment areas, and the carbon reduction (percentage decrease) was 37.1 (48.4%), 42.0 (41.7%), 48.9 (67.6%), 62.7 (70.0%), and 57.7 (66.5%) Mg C ha⁻¹, respectively.

The results above indicate that the implementation of forest projects leads to a significant decrease in carbon storage across vegetation and soil. In particular, depending on the intensity of the forest project, the carbon storage in aboveground vegetation decreased by 58.0%–85.8%, while soil carbon storage declined by 20.0%–29.8% due to physical disturbance. Given that degraded ecosystems require a substantial amount of time to recover to their original state, research on ecosystem resilience should be actively pursued.

Carbon-biodiversity linkage assessment

As displayed in Figure 6, carbon stocks and biodiversity changes after reforestation were expressed on a 5-point

scale for the carbon-biodiversity linkage assessment. Results showed that biodiversity differences were larger at Site 1 and Site 5 (–4.3), while Sites 2–4 showed smaller decreases (0.4 to –2.4). All sites demonstrated a carbon decrease of between –2.0 and 2.8 points, with the largest decrease at Site 5 and the smallest decrease at Site 2. However, there was no significant correlation between carbon stocks and loss of biodiversity.

Ranging from –2.2 to 7.1, the carbon-biodiversity linkage assessment showed that all sites had lower scores after logging (Fig. 7). Similar to the biodiversity results, Site 5 recorded the lowest value, while Site 4 recorded the highest score. The similarity to biodiversity is noted because the variation of carbon storage by region is small (± 0.3), while the variation of biodiversity is large (± 2.0) between high and low sites. As a result, Site 5, which experienced the largest decrease in carbon stocks and biodiversity due to the forestry project, also experienced the largest decrease in the linkage assessment index. According to the decrease

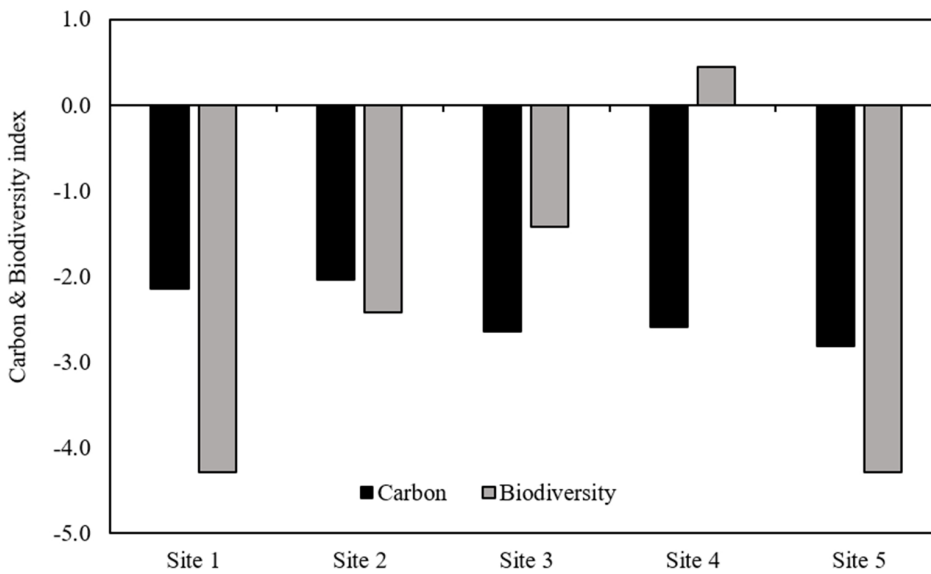


Fig. 6 Changes in carbon & biodiversity in each site due to forestry projects.

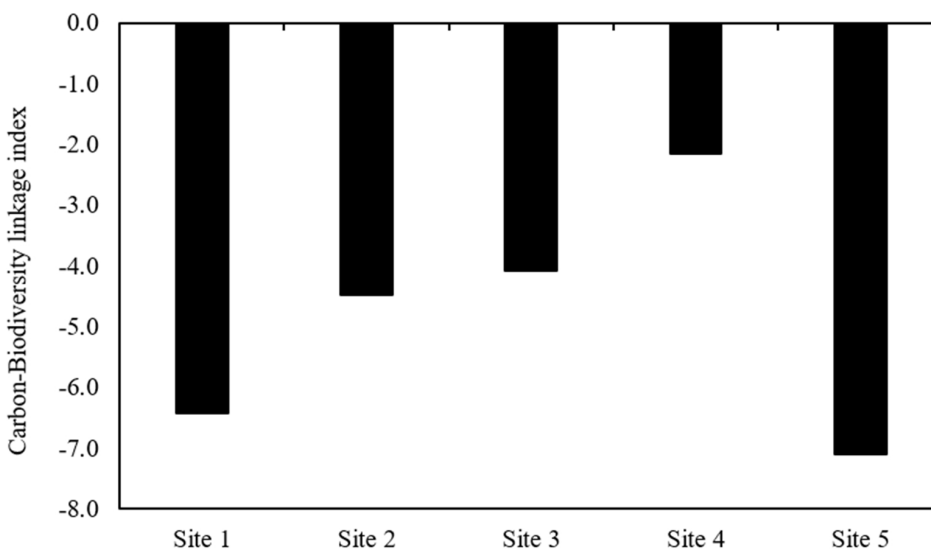


Fig. 7 Changes in carbon-biodiversity linkage index in each site due to forestry projects.

in carbon stocks and biodiversity, the remaining sites decreased in the order of Sites 1–4, with Site 4 having the smallest decrease in the linkage assessment index because of the offset between the negative carbon stocks and positive biodiversity.

Discussion

Biodiversity changes for each taxa

Our study compared biodiversity data from the assessment area where the forestry projects were conducted to determine the impact of forestry projects on biodiversity. In all plots, the results showed a biodiversity decline for most taxonomic groups. These findings are similar to other studies that have previously shown that biodiversity of terrestrial insects, mammals, amphibians, and reptiles is affected by forestry projects.

In most regions, insects have demonstrated a trend of declining biodiversity. It has been reported that, due to increased air and soil temperatures and reduced vegetation, beetles are negatively affected by the removal of woody vegetation (Barahona-Segovia et al. 2022; Česonienė et al. 2019). In addition, it has also been reported that native ants decrease and invasive species increase due to reforestation, affecting biodiversity (Zettler et al. 2004). However, insect biodiversity increased after the forestry project at Site 2. This result was similar to that of a study that conducted analysis on the role of forest thinning in the formation of high-diversity butterfly communities (Lee and Kwon 2014). In this study, forest thinning transformed vegetation to grassland, providing food for butterfly larvae as well as mating sites for adults (Lee and Kwon 2014). Furthermore, in Canada and Finland, beetle species richness and abundance were higher in logged areas because species with open habitat preferences increased as a result of logging, while generalist species were not dramatically affected (Noel et al. 1986). Thus, it appears that changes in vegetation and the resulting food sources and microenvironments affect insect biodiversity. Watt et al. (1997) found that certain groups, such as ants, Diptera, and arachnids, were more abundant in forest thinning areas than in clearcuts. Also, Lee et al. (2020) found that the abundance of Coleoptera and Lepidoptera increased significantly after thinning in Korean pine forests. These characteristics were found to be related to the increase in understory vegetation due to thinning. As such, it is believed that biodiversity is affected by insect preferences according to changes in the physical habitat of forests. In cases with outcomes such as Site 2 presented in this study, further research is required to characterize the relationship between vegetation, microenvironment, and dominant insects after forestry project conclusions. Conversely, analysis of the insect communities investigated after thinning has revealed effects of thinning

lasting for up to 10 years, so it takes a long time to restore the original biodiversity (Lee et al. 2020).

Inhabiting forests in south-central Chile, long-haired akodonts (*Abrothrix longipillis*) were reduced by at least 52.4% by logging activities and had higher predicted mortality rates due to effects such as increased exposure to predators and land cover change (Escobar et al. 2015). These results are similar to our findings in mammal biodiversity. However, and contrary to our results, several other studies have found that small mammal biodiversity is generally unaffected or benefited by forest harvesting (Michał and Rafał 2014; Sullivan et al. 2021). Analysis also found that while clear cutting generally decreased mammal biodiversity, thinning, a method of preserving some trees, increased mammal species richness and diversity (Sullivan and Sullivan 2014). These findings suggest that forests play an important role in maintaining the diversity of functions and characteristics of forests by providing the structural habitat complexity needed by forest mammals. Thus, an environment is created in which a wide range of organisms can survive and thrive after harvesting. However, variations in biodiversity are also likely to occur depending on the habitat characteristics of each mammal. For instance, aerial species such as flying squirrels (*Pteromys volans*) have been shown to be more sensitive to disruption of habitat continuity due to logging (Michał and Rafał 2014).

A global study of 460 amphibian and reptile communities found a 15.46% decrease in biodiversity in forests disturbed by logging, burning, and other activities (Iglesias-Carrasco 2023). Compared to other taxa in this study, amphibian and reptile biodiversity decreased significantly by -27.9 and -24.0 , respectively. This is attributed to the fact that amphibians and reptiles are generally less mobile and ectothermic animals, making them both vulnerable to habitat change (Gibbon et al. 2000). Amphibians utilize both terrestrial and aquatic ecosystems during their life cycle, migrating from their overwintering areas to the water's edge during the breeding season. Then, after breeding, they utilize these habitats for feeding and resting before returning to the overwintering areas (Lamoureux and Madison 1999; Lamoureux et al. 2002). Reptiles, like amphibians, survive by seasonally migrating between foraging, breeding, and overwintering areas to select suitable habitats throughout the year (Southwood and Avens 2010). Water balance and thermoregulation for both taxa can be disrupted by habitat disturbance from forestry projects, causing off-site movements (Sinch 1990). In this study, when compared to reptiles, amphibians showed higher biodiversity declines. This is consistent with a study that found that amphibians are more forest-dependent than reptiles, resulting in greater biodiversity loss from forest disturbance (Fulgence et al. 2022).

Avian biodiversity was also reduced by forestry projects. This is a result that was consistent with the decrease in

biodiversity found following clear-cutting of Brutian pine (*Pinus brutia* Ten.) forests in the Western Mediterranean region (Akdemir and Özdemir 2015). Similar results have been found in other studies of forestry impacts on bird biodiversity, including a study of the effects of reforestation on bird biodiversity in Estonia, Northern Europe, a study of bird diversity declines due to primary forest degradation and logging activities, and a metadata analysis of forestry-induced bird diversity declines (Bohada-Murillo et al. 2020; Löhms 2022; McCarthy 2012).

Benthic invertebrate biodiversity has been reduced by forestry projects, with Noel et al. (1986) and Kovalik (2018) reporting that biological communities are affected by reforestation due to alterations in habitat conditions such as light and temperature. Moreover, a comparison of lakes with and without logging showed decreased trout catches after logging, suggesting that logging has altered biodiversity (Bérubé and Lévesque 1998). An examination of the freshwater fish distribution in 2,129 watersheds in the United States analyzed distinct patterns of freshwater fish response to forests. The highest abundance and species were associated with forests and endangered fish were distributed according to the percentage of forest cover (Bury et al. 2021). After logging, dissolved organic carbon (DOC) concentrations and emissions in forest runoff increased significantly (Dahl 2008). By removing vegetation, logging exposes soil and organic matter is then washed into streams by precipitation. Stream water quality, habitat, and food webs are altered by these sediments, affecting the food resources for aquatic life. Reduced shade from logging has been reported to increase stream water temperatures, which subsequently causes changes in water quality, which stresses stream life and has long-term effects on species diversity (Erdozain et al. 2022).

Changes in carbon storage

In the study areas (sites 1–5), we found that, after forestry projects, above- and below-ground carbon stocks decreased. Previous studies have shown that above-ground biomass decreased by more than 68% after logging in Japanese larch (*Larix kaempferi*) and Korean pine (*P. koraiensis*) forests (Wang and Kim 2022), and forest biomass decreased by 48.8 and 40.8 Mg ha⁻¹, respectively. Further, by simulating 100 years of forest development, carbon storage was estimated in U.S. hardwood-conifer forests and found that reducing the frequency of logging and enhancing structural retention of forests were most effective in increasing carbon stocks (Schwenk et al. 2012).

Various studies have reported decreases in soil carbon stocks after forestry projects. Kim (2008) found a decrease in soil organic carbon content from 8.06% to 9.12% after clear-cutting in Hamyang-gun, South Korea. Moreover, a decrease in soil organic content from 4.77% to 1.65% was found after clear-cutting in a black alder forest (*Alnus glu-*

tinosa) in northeastern Turkey (Yüksek and Yüksek 2009). After forestry projects, the decrease in soil organic matter suggests that soil carbon in physically disturbed forests after clear-cutting is being released by precipitation and soil respiration. Kim (2008) also found that moisture and carbon storage in forest soils decreased, temperature and soil respiration increased, and soil physicochemical properties changed after clear-cutting. Furthermore, a simulation study of the effects of clear-cutting on soil erosion and runoff found that soil erosion increased more after clear-cutting (Dung and Kim 2021). Mills et al. (2023) found that carbon was released from clear-cut forest ecosystems in Borneo's tropical forests, and predicted that this trend would continue for at least another decade.

Forest ecosystems have received much attention for their importance in climate change mitigation. Research is being conducted on changes in land use to increase forest cover and avoid reforestation, as well as carbon management and existing forest conservation (McKinley et al. 2011). Ameray et al. (2021) concluded that primary forest conservation can contribute to climate change mitigation by increasing soil carbon stocks, thereby removing atmospheric GHGs. Noormets et al. (2015) compared the carbon flux of managed and unmanaged forests and concluded that efforts to maximize productivity in managed forests can have a negative impact on soil carbon storage. Therefore, forest management strategies should not simply focus on resource production. The multiple processes involved in the carbon cycle should be taken into consideration to simultaneously promote long-term carbon sequestration and ecosystem health. Overall, it is known that, in forests, more carbon is stored in the soil than in vegetation (Fahey et al. 2010). Previous studies have shown that Korean forests, including national parks, have a large amount of carbon stored in their vegetation, and these forest soils are expected to have high carbon storage capacity (Jeong et al. 2023). Therefore, forests with high conservation value, such as virgin forests, should be actively protected to maintain the carbon storage capacity of vegetation and soils in the long term. Land use policies, such as the expansion of protected areas, should be developed to minimize logging and conserve biodiversity.

Carbon-biodiversity linkage assessment

In recent years, there has been a growing body of research on the interaction between carbon stocks and biodiversity around the world. A strong correlation between areas of high carbon storage and high biodiversity was found by Strassburg et al. (2010). However, weaker correlations were shown between endangered species and species with restricted ranges, based on a sample size of 12,500 km², the analysis may not be accurate for smaller areas. Soto-Navarro et al. (2020) produced a high-resolution map of carbon stocks and biodiversity around the world. By overlaying carbon stocks and biodiversity on the map, hotspots were

identified. However, due to the aggregated data obtained from a wide range of regions, it is difficult to reflect the characteristics of micro-regions and predict dynamic changes over time. Thus, the ability to predict changes in carbon stocks and biodiversity is limited due to climate change or changes in human activities. Sabatini et al. (2019) examined the relationship between carbon stocks and biodiversity in temperate forests of Central Europe. The relationship was found to be weak and highly variable, suggesting that carbon stocks and biodiversity vary by region and that management strategies are needed to optimize co-benefits. Vijay et al. (2022) examined the co-benefits of land protection policies on biodiversity and ecosystem services in the continental United States. They found a high correlation between carbon stocks and biodiversity, suggesting that areas which are simultaneously rich in biodiversity and high carbon storage capacity have the potential to provide multiple ecosystem services. Conversely, the study suggests that the benefits of biodiversity and carbon storage are harder to optimize in densely populated areas, resulting in weaker ecosystem services such as recreation. The study also found that the spatial overlap between carbon stocks and biodiversity is not always consistent, and that the results may not fully reflect the effects of climate change and land use change over time.

Based on the aforementioned studies, carbon-biodiversity linkage assessments have demonstrated either strong or weak correlations, likely due to large regional variability. In addition, linkage assessments for micro-regions are uncertain, especially at large regional scales. The correlation between carbon stocks and biodiversity was not found in our study areas (Sites 1–5), thus indicating a similar trend to the results of previous studies. Furthermore, the total area of each site is about 154.84 km², and it is difficult to explain the carbon-biodiversity characteristics of smaller forests, wetlands, grasslands, etc. A limitation of this study is that it utilized multiple ecological information (vegetation, soil, species information, etc.) at the same time in a specific area. Previous studies have demonstrated that spatial and temporal data inconsistencies can increase uncertainty in the linkage assessment results. Therefore, to compensate for this, it is necessary to build a systematic database from the basic data generation stage to continuous management.

Carbon-biodiversity linkage assessments could play a key role in the implementation of environmental policies to mitigate climate change in the future. Soto-Navarro et al. (2020) identified hotspots where both carbon stocks and biodiversity are high. These hotspots offer the potential to simultaneously increase biodiversity and carbon storage capacity and can support policy decisions as priority areas for conservation. Lecina-Diaz et al. (2018) also referred to forest areas with high carbon stocks and biodiversity as hotspots, noting that these areas play an important role in

ecosystem services and biodiversity conservation. Evidence base for management strategies and policies to maximize and conserve ecosystem services in hotspots. O'Brien et al. (2023) suggested that high carbon storage capacity areas may also be important for biodiversity conservation, noting that biodiversity is particularly high in carbon-rich boreal forests and peatlands. Through the integration of conservation strategies around these hotspots, national biodiversity and climate goals could be met. To support policies to address climate change mitigation and adaptation, Korea should further improve the scientific methodology for identifying hotspots and continue to conduct fundamental research on the trade-offs and synergies between carbon stocks and biodiversity.

Conclusions

This study analyzed the impact of forestry activities on carbon stocks and biodiversity. The results indicate that forestry activities, including reforestation, have a negative impact on both carbon stocks and biodiversity. In particular, aboveground and soil carbon stocks have been significantly reduced, and biodiversity has declined for many taxa. Forestry operations have altered the structure of forests, resulting in habitat destruction, which in turn has caused species habitat degradation. While our study did not find a direct correlation between carbon stocks and biodiversity decline, there were distinct pre- and post-project changes in each region. These findings emphasize the need to maintain and enhance ecosystem functioning when formulating forest management and conservation policies. Future forestry projects should consider both biodiversity and carbon storage capacity and be conducted in a sustainable manner. Moreover, future policies should identify and protect hotspots with both high carbon stocks and biodiversity, which could play a key role in climate change mitigation and enhancement of ecosystem services.

Abbreviations

BD:	Bulk density
BEF:	Biomass expansion factor
C:	Carbon content
CaF:	Carbon fraction
CoF:	Coarse fragment
DBH:	Dimeter at breast height
NES:	Natural environment survey
T:	Soil depth
V:	Volume
WD:	Wood density

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Authors' contributions

Conceptualization: H.M.J., I.J., S.R.K.; Data curation: H.M.J., S.H., C.H.C., I.J.; Formal analysis: H.M.J., S.H., C.H.C., I.J.; Investigation: H.M.J., S.H., C.H.C., I.J.; Methodology: H.M.J., S.H., C.H.C., I.J., S.R.K.; Project administration: S.R.K.; Software: H.M.J., S.H., C.H.C., I.J.; Supervision: S.R.K.; Validation: S.H., C.H.C., S.R.K.; Visualization: H.M.J., S.H., C.H.C.; Writing-original draft: H.M.J., I.J.; Writing-review & editing: I.J., S.R.K.

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Consent for publication

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Competing interests

The authors declare that they have no competing interests.

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