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Spatiotemporal assessment of deforestation and forest degradation indicates spillover effects from mining activities and related biodiversity offsets in Madagascar

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ABSTRACT

Mining has severe environmental and social impacts. To compensate for the environmental damage caused at mining sites, mining companies are required to engage in biodiversity offsetting activities elsewhere. In forest landscapes, most offsetting policies focus on compensating for biodiversity loss from deforestation, while forest degradation is largely ignored – even though it contributes substantially to biodiversity loss. One reason for this is that forest degradation is challenging to assess and monitor. This study focuses on a large nickel and cobalt mine in Madagascar. By analysing remote sensing time series, we assess detailed annual forest change dynamics and distinguish different types of forest disturbance within and around the mining lease area and the two main associated biodiversity offset areas between 2006 and 2020. Our results show that deforestation rates within the two biodiversity offset areas are low (18 ha, or 0.4%; 164 ha, or 2.4%), suggesting that conservation measures are effective. However, this is not the case when looking at forest degradation. We found that substantial shares of forest within the two biodiversity offset areas are affected by degradation (545 ha, or 11.4%; 662 ha, or 9.7%). In the surrounding unprotected landscape, the rates of deforestation (451 ha, or 6.7%; 553 ha, or 4.9%) and forest degradation (2360 ha, or 34.8%; 5794 ha, or 51.1%) are much higher. The spatiotemporal pattern indicates spillover effects for both deforestation and forest degradation. Taken together, our findings show that restrictions on local communities' access to forest resources within biodiversity offset areas affect the surrounding landscape and can cause substantial additional adverse environmental impacts there. We also demonstrate that forest degradation monitoring is feasible, and that forest degradation is widespread even though it is still largely ignored. These findings should be considered in future biodiversity offsetting policies and best practices.

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1. Introduction

Anthropogenic and natural forest disturbances cause ecological damage and carbon emissions. The degradation of forests in developing countries, particularly in tropical and subtropical latitudes, is considered a major contributor to global greenhouse gas emissions (Pearson et al., 2017). While national commitments to reduce emissions resulting from land use change and land management mostly focus on curbing deforestation, the role of forest degradation is still largely neglected in policy discussions (Silva Junior et al., 2021) and rarely considered in conservation initiatives (e.g. Reducing Emissions from Deforestation and forest Degradation, also known as REDD+) (Mertz et al., 2012; Panfil and Harvey, 2016). At the same time, it is known that human-induced forest degradation caused by fires, selective logging, and edge effects can result in similarly large carbon dioxide (CO_2) emissions (Aragão et al., 2018; Silva Junior et al., 2022) and is a major driver of socio-environmental impoverishment (Barlow et al., 2016; Brando et al., 2020; Bustamante et al., 2016; Caviglia-Harris et al., 2016; Geist and Lambin, 2003; Miyamoto, 2020).

Today's remaining tropical forests face many anthropogenic threats, including mining. The potential of mining as a means of national economic development presents a dilemma when weighed against the likelihood of environmental destruction. This is particularly true for countries in the global South that are rich in mineral resources and biodiversity. On the one hand, mining can be a vital source of revenue for governments, contributing to economic growth, generating employment opportunities, and stimulating local economies (Ericsson and Löf, 2019; McMahon and Moreira, 2014). On the other hand, mining is considered a significant driver of deforestation and forest degradation that acts both directly (e.g. construction of roads, mining sites, and settlements) and indirectly (e.g. opening up previously inaccessible areas to illegal logging or agricultural expansion) (e.g. Asner and Tupayachi, 2016; Giljum et al., 2022; A. Mishra et al., 2022a; Sonter et al., 2018, 2017). Extractive industries such as mining often require clearing large areas of forest to access underground mineral resources. This process involves the removal of vegetation and the disruption of ecosystems, leading to loss of biodiversity and destruction of habitats for many plant and animal species (Armendáriz-Villegas et al., 2015; Deikumah et al., 2014; Murguía et al., 2016; Siqueira-Gay et al., 2020; Sonter et al., 2020; Thakur et al., 2022). Additionally, mining operations can lead to soil erosion, sedimentation of water bodies, and contamination of rivers and streams, further affecting surrounding forests and ecosystems (Byrne et al., 2012).

One of the conditions that mining companies must fulfil to obtain a mining lease and permit as well as funding from international donors is that they comply with international standards promoting the minimization of negative environmental and social impacts. They must compensate for the unavoidable, irreparable damage caused by deforestation, and one way of doing this is to offset deforestation and biodiversity loss elsewhere (Bull et al., 2013; Zu Ermgassen et al., 2019). However, national and international policies to minimize negative environmental impacts of commercial land uses often focus exclusively on avoiding deforestation while leaving forest degradation unconsidered (Silva Junior et al., 2022). Furthermore, deforestation and forest degradation might not only occur on site; it might also be caused elsewhere via so-called leakage or spillover effects of compensation measures, namely indirect land use changes and an acceleration of deforestation and forest degradation due to increased pressure on openly accessible forest patches outside of newly established protected areas (Meyfroidt et al., 2020). Such unintended (indirect) consequences of land leases and related conservation policy interventions and certification schemes have been reported in various geographical contexts (Heilmayr et al., 2020; Magliocca et al., 2020; Zaehringer et al., 2018).

Madagascar is known to have deposits of various valuable minerals, including nickel, cobalt, chromite, ilmenite, and graphite (Yager, 2017). All of them are highly important for the ongoing energy transition. Accordingly, the mining sector in Madagascar has the potential to contribute to the country's economic development by generating tax revenue, creating employment opportunities, and attracting foreign investment. However, mining activities in Madagascar also pose significant challenges and raise environmental and social concerns. One of the main issues is the potential impact on the country's unique and biodiverse ecosystems with their numerous endemic plant and animal species (Mittermeier, 1988). Madagascar has been experiencing continued deforestation in the 21st century (Suzzi-Simmons, 2023). It is driven by a combination of socio-economic, political, and environmental factors. Important reasons include a high dependence of the population on forest resources for their livelihoods (i.e. for firewood, wild foods, charcoal production, shifting cultivation agriculture, pasture creation) (Suzzi-Simmons, 2023), illegal selective logging (Allnutt et al., 2013; Randriamalala and Liu, 2010), commercial agriculture (Vieilledent et al., 2020), urbanization, mining, climate change (Hending et al., 2022), and weak governance and enforcement (Horning, 2018). Deforestation and forest degradation are considered to be among the most significant threats to terrestrial biodiversity in Madagascar (FAO and UNEP, 2020; Ralimanana et al., 2022). Mining activities are likely to further exacerbate that threat.

One example is the Ambatovy mine, one of the world's largest open-pit nickel mines (Mining Technology, 2023). It is situated in the middle of a highly biodiverse rainforest, surrounded by several conservation areas, and located in proximity to two national parks (Berner et al., 2009). To offset biodiversity loss from deforestation caused by the implementation of the mine, the Ambatovy company has identified four biodiversity offset areas in the region, in which it aims to reduce deforestation; all four are in areas where deforestation is mostly driven by small-scale agriculture (Berner et al., 2009; Devenish et al., 2022). Results obtained by Devenish et al. (2022) suggest that Ambatovy's efforts to slow deforestation in the defined biodiversity offset areas are effectively averting deforestation and compensating for the forest loss caused by the mine. At the same time, scientific evidence on the socio-economic impacts of Ambatovy's activities shows that the combination of the long-term mining lease and protection of large parts of the surrounding natural forests is impeding access to forest resources for local communities and, more importantly, increasing the pressure on land surrounding the areas protected by restrictions (Bidaud et al., 2017). This has increased social tension and conflicts in communities, with substantial negative impacts on well-being. However, no study to date has shed light on potential forest degradation effects of the mine, and no research has been done on off-site effects of the Ambatovy-funded conservation measures in terms of the possible displacement of deforestation or forest degradation into unprotected remaining forest patches.

Regardless of country or region, precise evidence on the continued loss and degradation of forests in the context of mining is key to establishing governance frameworks and policies capable of halting these processes and thereby reducing their contribution to climate change as well as preserving biodiversity and other ecosystem services. Satellite data are a useful and cost-efficient basis for generating spatial and temporal information on forest cover changes (Achard et al., 2007; De Sy et al., 2012; Hansen et al., 2013). However, while deforestation usually shows as an abrupt change in spectral reflectance in a satellite image and is relatively easily captured with freely available medium-resolution optical satellite data provided, for example, by Landsat, the more subtle changes of forest degradation are harder to detect. Forest degradation leads to changes in the three-dimensional canopy structure, such as canopy openings, changes in vertical foliar density, and changes in plant area density. In addition, disturbances are often small and the regeneration of the canopy cover is rapid, making the disturbances difficult to capture by means of satellite remote sensing (Gao et al., 2020; Milodowski et al., 2021); some of the structural changes and biomass loss occurring below the canopy may not be noticed from above. Nevertheless, many disturbances (e.g. selective logging, fires, shifting cultivation) result in substantial canopy thinning and gaps that cause a change in spectral reflectance, spectral endmember composition of pixels, and image texture (Eckert et al., 2011; Gao et al., 2020). Optical remote sensing approaches aim at capturing these degradation effects, for example, by combining time series and spectral mixture analysis, although the captured effects will likely only represent fractions of the total degradation taking place. Recently available cloud computing services have made it possible to quickly screen entire satellite data catalogues and filter out clouds, cloud shadows, and other atmospheric influences to generate cloud-free geospatial satellite data time series. Such a dataset then makes it possible to capture the dynamics of abrupt and subtle forest change processes and the specific point in time or period of their occurrence.

In this study, we adapted and applied a set of remote sensing time series analysis tools developed by Bullock (2019) to assess both deforestation and forest degradation between 2006 and 2020, capturing forest changes before, during, and after implementation of the Ambatovy mining lease. To single out forest degradation caused by shifting cultivation, we also capture multiple consecutive disturbances. The geographical focus is on Ambatovy's mining lease area and on two out of four biodiversity offset areas Ambatovy is using to compensate for forest loss caused by the mine. Furthermore, we analyse whether and how forest cover change dynamics and rates have changed in the surroundings of the mining lease area and the biodiversity offset areas, with the aim of understanding how the increasing restrictions on local communities' access to forest resources affect the surrounding mosaic landscape and its fragmented forest patches. We intend to demonstrate that forest degradation monitoring is feasible and can provide a more detailed spatiotemporal picture of the direct and indirect on- and off-site impacts of large-scale mining activities on tropical rainforests. Our aim is to inspire monitoring platform developers as well as inform policymakers.

2. Study area

The Ambatovy mine is one of the largest long-life lateritic nickel and cobalt mines in the world. It is located in eastern Madagascar, near the town of Moramanga. Construction took place between 2007 and 2011, and the mine became operational in 2012. Development of the mining site involved occupying agricultural land, resettling two households living within the future mine footprint, and barring access to culturally significant sites (Dynatec Corporation of Canada, 2006a). The mining site lies in an area formerly covered with biodiversity-rich zonal as well as a smaller patch of azonal tropical forest, and remains largely surrounded by forest today (Berner et al., 2009). In addition to the mining site, Ambatovy's mining infrastructure includes a slurry pipeline connecting the mining site with tailings facilities and dams on the east coast, near the town of Toamasina. The company also operates a processing plant in Toamasina with access to a port for shipping. The Ambatovy mining lease area consists of the open-pit mine and a surrounding biodiversity offset area known as "Conservation Zone" which was placed under protection in 2009. Ambatovy is responsible for preventing biodiversity loss in four biodiversity offset areas (Ambatovy, 2022; Hase et al., 2014) to offset the mining activities' (negative) biodiversity impacts: (1) the above-mentioned *Conservation Zone*, which directly surrounds the mine footprint (i.e. the area taken up by the open-pit mine) with two patches of rare azonal forest and consists mainly of zonal forest; (2) the *Ankerana Forest Massif*, an area consisting of azonal and zonal forest that was placed under protection in 2011 and is located about 70 km northeast of the mining site; (3) the *Torotorofotsy Ramsar Wetland* to the south of the lease area (protected since 2014); and (4) a larger, biodiversity-rich area immediately northeast of the mining lease area, which is located between two protected areas and has been proposed for protection, the *Corridor Forestier Analamay-Mantadia* (Fig. 1). Two other important protected areas exist near the mining site: the *Analamazaotra Special Reserve* and *Mantadia National Park*, which together form *Andasibe-Mantadia National Park* and are located about 17 km southeast and east of the mining lease area, respectively. The entire tropical forest massif of which all these protected areas are part is known as the *Corridor Ankeniheny-Zahamena*, which, in its entirety, is also a protected area that was established in 2015 (Hase et al., 2014; Ramahavalisoa et al., 2012; Ramsar Sites Information Service, 2016).

For the analysis of potential deforestation and forest degradation spillover effects in the immediate vicinity of the Ambatovy mine and its biodiversity offset areas, we defined two buffer areas with a radius of 3 km. This distance corresponds to the average agricultural activity range of the majority of the local population living in close proximity to the mining lease and the biodiversity offset areas. Furthermore, it aligns with the study area of a parallel study using household surveys in the villages within the buffer area of the mining lease (Zaehringer et al., 2024), as well as the study area of a previous study by Bidaud et al. (2017), who conducted interviews, focus group discussions, and a household survey in villages located in close proximity to the Ankerana Forest Massif. The first of the two buffers in our study surrounds the entire mining lease area and was selected because it contains the mine footprint and much of the Conservation Zone, as well as smaller parts of the *Corridor Forestier Analamay-Mantadia* and the *Torotorofotsy Ramsar Wetland*. Consequently, we did not analyse buffer areas around each of these individual biodiversity offset areas, because they would

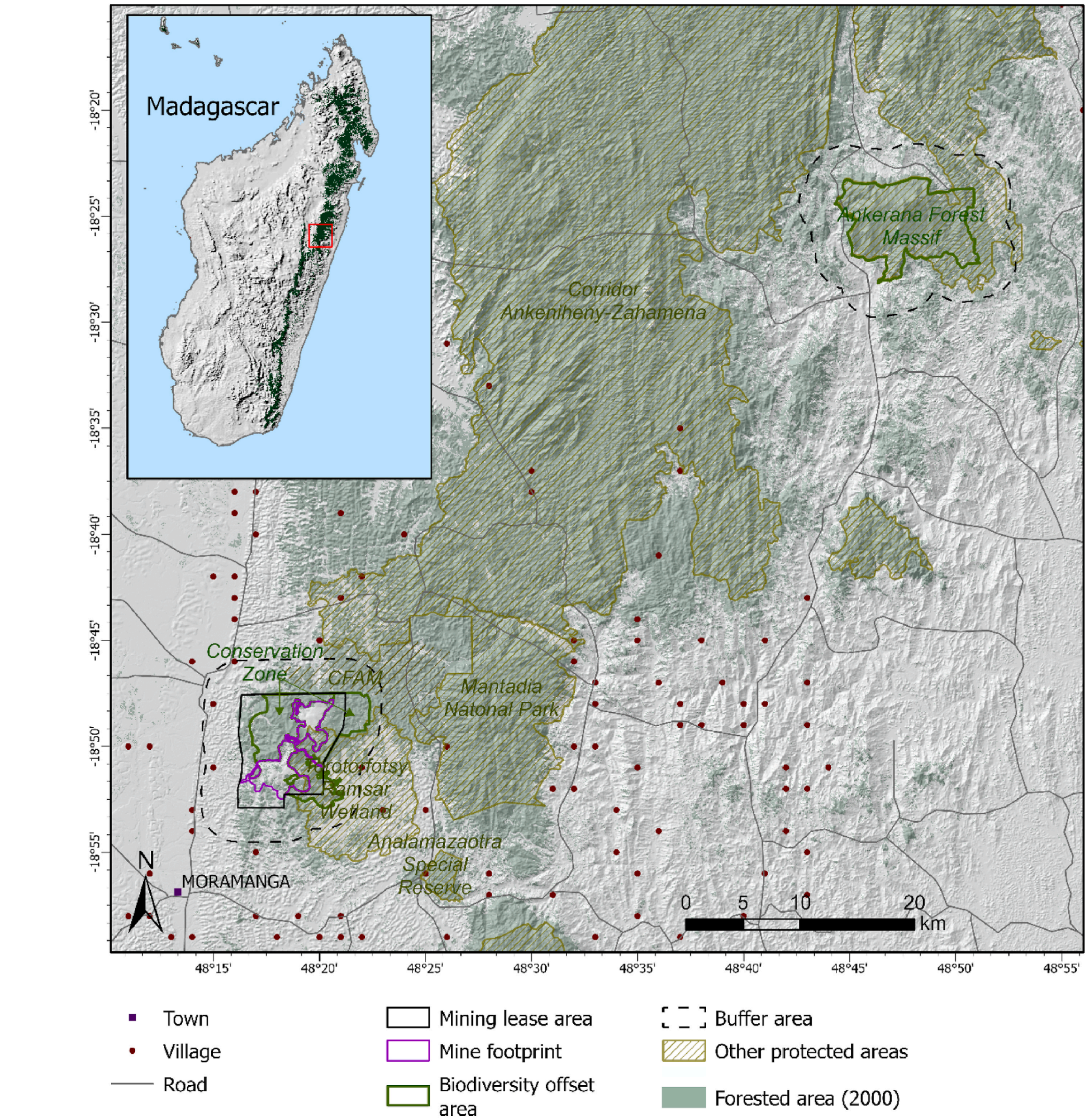

Fig. 1. Overview of the study area in Madagascar, showing the wider setting of the biodiversity offset areas, other protected areas, and the two 3-km buffer areas. CFAM, Corridor Forestier Analamay-Mantadia. Data sources: Forest area (2000), Hansen/UMD/Google/USGS/NASA (Hansen et al., 2013); other protected areas, Protected Areas of Madagascar (https://protectedareas.mg/); towns, The World Bank Data (https://data.worldbank.org/); villages, UN-OCHA; roads, Digital Chart of the World, DIVA-GIS (https://www.diva-gis.org/gdata); all other layers were created by the authors.

have included large parts of the mining lease area, the mine footprint, or adjacent protected areas. The second buffer area surrounds the Ankerana Forest Massif, which was declared a biodiversity offset area and placed under protection in 2011.

The mining site is located on the eroded remains of a plateau at about 1100 m a.s.l. The surface of the plateau is relatively uneven, with numerous depressions forming ephemeral pools. Several small streams have their source in the actively mined area and used to flow off from there. Before Ambatovy implemented their mining infrastructure, most of the mining site was covered with natural forest (Dynatec Corporation of Canada, 2006a); some parts of it had already been deforested or degraded as a result of earlier human-induced pressure (e.g. hunting, logging, fires, agriculture) (Ambatovy, 2019; Dynatec Corporation of Canada, 2006b; Reyes and Abhukara, 2015). The neighbouring Ramsar site was, and still is, largely covered by patches of primary and secondary forest, along with grassland, marsh herbaceous vegetation, eucalyptus plantations, woodlots, rice paddies, and slashed and burnt areas used for shifting cultivation (Ambatovy, 2019; Dynatec Corporation of Canada, 2006b).

The Ambatovy mine is the largest foreign investment in the country to date. It has an annual production capacity of 60,000 tonnes of refined nickel and 5600 tonnes of cobalt (Ambatovy, 2021). It received environmental and exploitation permits in 2006, authorizing the company to extract nickel, cobalt, copper, platinum, and chromium for a period of 40 years (Elias et al., 2018). The mine has been in operation since 2012, with an estimated total operating life of 29 years. It was established by an international joint venture and has received debt financing from various lenders, including state-sponsored export credit agencies, international development banks, such as the European Investment Bank, and commercial banks from around the world (Ambatovy, 2021; EIB Complaints Mechanism, 2018). The project is subject to the 1999 *Malagasy Mining Code* (Government of Madagascar, 2005a) and the national *Law on Large Investments (LGIM)* dating back to 2002 (EITI, 2019; Government of Madagascar, 2005b).

3. Methods

3.1. Forest disturbance detection

To detect forest cover changes in the four selected management zones – the mining lease area, the mine footprint, the Conservation Zone, and the Ankerana Forest Massif – as well as the two defined buffer areas, we adapted and applied the Continuous Degradation Detection (CODED) algorithm developed by Bullock (2019). CODED is implemented in Google Earth Engine (GEE). Based on Landsat satellite time series analysis, CODED enables repeated mapping and estimation of the spatial distribution of forest change over large areas and simultaneously at a subpixel level. Its ability to detect subtle and small-scale forest changes and to distinguish between different types of forest disturbance has been demonstrated in several studies (Aryal et al., 2021; Bullock et al., 2020a, 2020c, 2020d). The methodological workflow presented in the following is illustrated in Fig. 2.

Generally, challenges in the analysis of Landsat optical satellite data arise from clouds, shadows, haze, atmospheric influences, and data gaps. All of these effects can contribute to erroneous identification of forest cover changes (Bullock et al., 2020c; Pimple et al., 2017; Zhu and Woodcock, 2012). To limit the influence of these factors, all available Landsat Collection surface reflectance images available for the study area and period were filtered for clouds and cloud shadows. The annual availability of filtered Landsat imagery suitable for analysis is visualized in Fig. S1 in the supplementary data file.

Detecting forest degradation events from satellite imagery is inherently difficult (Bullock et al., 2020c; Herold, 2011; Herold et al., 2011), especially when they occur at subpixel scales (< 30 m; e.g., from selective logging, cf. Allnutt et al. (2013)). To reliably detect these small-scale disturbance events, CODED uses subpixel spectral mixture analysis (SMA) to calculate and evaluate time series changes in the Normalized Difference Fraction Index (NDFI) (Souza and Barreto, 2000; Souza and Roberts, 2005). SMA assumes that each pixel consists of different proportions of spectrally pure end members. These proportions differ in terms of the structure and composition of physical elements in the analysed pixel. The satellite images are converted to the end members *green vegetation* (GV), *non-photosynthetic vegetation* (NPV), *shadow*, and *soil*. Then, the different end member fraction images are transformed into the Normalized Difference Fraction Index (NDFI) based on the methodology developed by Souza and Roberts (2005; 2013) and adapted by Bullock et al. (2020c). See equations (4) and (5) in Souza and Roberts (2005) for the calculation of the NDFI. NDFI values are low for pixels with high proportions of soil and NPV. Accordingly, the magnitude of change in NDFI in a time series dataset is assumed to be a suitable proxy for the extent of forest canopy damage or tree removal and can serve to detect possible forest degradation (Bullock et al., 2020a; Souza and Barreto, 2000). CODED uses a regression-based structural break test on the time series of each pixel to detect changes in vegetation cover. For this purpose, the end member fractions and NDFI data time series are fitted to ordinary least square

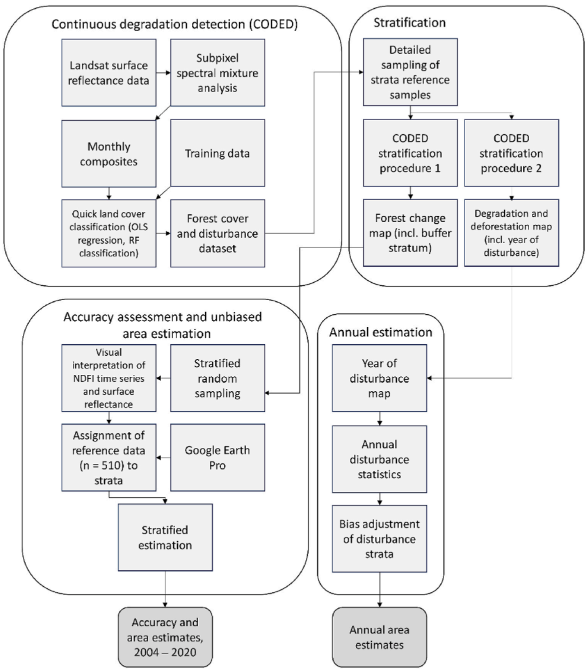

Fig. 2. Methodological workflow, modified and adapted from Bullock et al. (2020d).

(OLS) regression models for a defined reference (i.e. training) period (in this study, the three years from 2017 to 2019). To reduce processing time and to distinguish forest disturbances and deforestation from other permanent land cover changes, CODED then performs a quick random land cover classification and generates a forest cover mask. The OLS intercept, the regression coefficients, and the model's root mean squared error (RMSE) serve as inputs for this classification (Bullock et al., 2020b, 2020c). We trained the random forest algorithm using 767 training samples representing the land cover types of forest, non-forest vegetation (i.e. grass, crops, shrubs), water, and unvegetated or bare land (i.e. burnt, ploughed fields, roads, buildings).

Several parameters control the sensitivity of CODED to forest changes. Their tuning is particularly important for the detection of subtle changes such as forest degradation. We started with the input parameters developed for the analysis of rainforest in the Amazon (Bullock et al., 2020c) and adjusted them iteratively to the Malagasy rainforest in our own study area. In the end, we retained all parameters except *thresh* and *consec*. *Consec* defines the number of consecutive (monthly) observations that must show a change value lower than the normalized threshold (*thresh*) for the NDFI in order for the algorithm to detect a forest disturbance. The particularly high cloud coverage in Malagasy rainforests and the general lack of Landsat satellite data for certain years (Fig. S1) frequently meant that fewer than five clear consecutive observations were available for an entire year. This made it necessary to adjust the *consec* and *thresh* parameters, as the algorithm would otherwise have missed important short-term as well as subtle (low-magnitude) forest disturbances. We reduced *consec* from five to three consecutive monthly observations and set *thresh* from 5.0 to a normalized NDFI value of 4.0. All parameter settings can be found in Table S1 in the supplementary data file.

Once all parameters are defined, the CODED algorithm is run for the study period (2004–2020 in our case). Based on the random-forest-based forest cover mask, a first regression model is fitted to the NDFI and its end member fractions. Once the algorithm has detected a disturbance, a second, refined regression model is fitted to the NDFI data, but skipping one year after the disturbance to avoid classification errors in a potentially recovering degraded forest. In this study, we specified a minimum interval of three years between disturbances to account for the time needed for the vegetation to regrow to the point where a subsequent disturbance could potentially occur. Accordingly, reliable forest disturbance results are obtained for 2006 onwards. Finally, differences in post-disturbance land cover recovery characteristics, calculated from the NDFI regression coefficients, are used to distinguish between deforestation and degradation events (Bullock et al., 2020c). The resulting CODED disturbance dataset consists of five vectors containing the following information for each pixel location: the change date (of the first and any subsequent disturbances), the change magnitude, the post-change land cover class (i.e. forest or non-forest), a forest/non-forest class flag (corresponding to the initial training data label), and the relative NDFI difference (i.e. the relative magnitude of the post-disturbance NDFI compared to the pre-disturbance NDFI, in per cent) (Bullock et al., 2020b). These five vectors serve as input for a second, more detailed sampling of statistically representative so-called strata reference samples, which are then used for the CODED stratification procedure and to generate a detailed forest change map. This forest change map serves as the basis for an accuracy assessment and for the calculation of accurate areal change estimates. Collecting representative strata samples is particularly important for forest disturbance classes, as these cover a relatively small area compared to the size of the entire study area (Bullock et al., 2020c; Olofsson et al., 2014). In this study, we defined two stable strata, (1) forest and (2) non-forest; three change strata, namely (3) degradation, (4) deforestation, and (5) multiple consecutive disturbances (including rotational and other regrowth and disturbance, and degradation and consecutive deforestation, which were merged due to high confusion rates); and a (6) buffer stratum. The latter was introduced to identify areas surrounding disturbance events in forest areas. This was based on the idea that a disturbance event detected for a given pixel will most likely have caused disturbance in the surrounding pixels, even if this remained undetected by the algorithm. Pixels in the buffer stratum show an unclear vector time series characteristic and are extremely hard to accurately assign to the correct disturbance category. Accordingly, we excluded these buffer pixels from being assigned to one of the disturbance strata to achieve a more accurate and reliable forest change estimate. The additional multiple-disturbance stratum was introduced to account for human-induced consecutive forest disturbance patterns typical of the study area, making it possible, for example, to capture rotational shifting cultivation activities and separate them from other factors causing forest degradation. Fig. 3 illustrates the characteristic NDFI curves for each of the defined strata.

3.2. Accuracy assessment and unbiased area estimation

To assess map accuracy and calculate more reliable estimates of the strata's area shares, a stratified random (reference) sampling was performed based on the stratified map. The stratified random sampling design is particularly suitable for categorical observations, as it offers the option to increase the sample size of strata with small areal proportions (change strata) compared to other (stable) strata. This reduces the standard error in the accuracy estimation of each selected stratum (Arévalo et al., 2020b; Olofsson et al., 2014). The reference sample size was calculated using Cochran's sample size formula (1977, Eq. (5.25), p. 98), adapted by Olofsson et al. (2014) and implemented in GEE. We used the AREA2 tool, implemented in GEE (Bullock et al., 2020a), to collect reference samples. This was done through visual interpretation of NDFI time series and corresponding surface reflectances derived from Landsat satellite data. For each time step, samples were visually cross-checked on Google Earth Pro historical satellite imagery to assign the correct stratum to each of the 510 randomly selected reference samples. This assignment was done independently of the above-described stratification process in order to avoid any interpretation bias (Olofsson et al., 2014).

The reference samples were then compared with the CODED stratification result using an unbiased stratified estimator (Arévalo et al., 2020a; Bullock et al., 2020c), based on the methodology of Olofsson et al. (2014) and Stehman (2013). The estimator is required to estimate population parameters from sample data generated in the stratified random sample (Bullock et al., 2020b). The combined outputs from the automated stratification and the assigned reference data are used to calculate error matrices, stratum area shares and stratum area estimates (including standard errors and 95% confidence intervals), as well as user's, producer's, and overall accuracy.

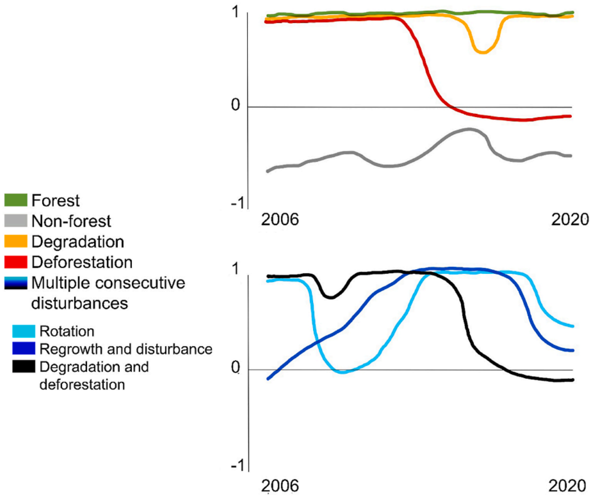

Fig. 3. Freehand sketch of NDFI time series curves that are typical of the specific strata.

Finally, we compared our degradation and deforestation classifications with the corresponding datasets recently published by Vancutsem et al. (2021). These datasets were developed specifically for tropical forest regions around the globe and have been thoroughly validated. To calculate omission and commission errors, we first aggregated the annually resolved datasets and then compared our classifications with them irrespective of the year in which deforestation or degradation had occurred.

3.3. Annual deforestation and forest degradation estimation

In addition to the bitemporal CODED analysis assessing forest cover change between 2006 and 2020, we conducted an analysis of annual forest change. For this purpose, we performed a second CODED stratification analysis, considering only the two disturbance strata of Degradation and Deforestation (i.e. just one disturbance event) and following the same classification rules we had previously defined for these strata. We modified the CODED code in GEE to write the year of identified disturbance to an additional, sixth output dataset, resulting in a spatial map of years in which a pixel became degraded or deforested. This was then used to calculate the annual area shares of observed deforestation and degradation.

The annual areal shares for each of the two disturbance strata (i.e. deforestation and degradation) were adjusted by multiplying them with the quotient between the area-weighted stratified areal estimation and the unadjusted areal share. Assuming that the bias is evenly distributed over time and space, we used it to recalculate the annual disturbance area shares for the selected strata.

4. Results

4.1. Observed deforestation and forest degradation between 2006 and 2020 in the studied management zones and buffer areas

Between 2006 and 2020, the mine footprint experienced the highest share of deforestation (63%, 920 ha), caused by the implementation of the active mining site (Fig. 4). Another 27% (395 ha) of forest experienced degradation or multiple consecutive disturbances. This adds up to a loss of 90% (1315 ha) of the natural forest that formerly grew in the mine footprint area (see also Table 1). The Conservation Zone adjacent to the mine and the Ankerana Forest Massif (hereafter referred to as Ankerana) experienced much less disturbance, and this consists mostly of forest degradation – 12% (1216 ha) in the Conservation Zone and 9.7% (662 ha) in Ankerana – with an additional small share of deforestation in Ankerana (2.4%; 164 ha). In total, the official mining lease area – which includes the mine footprint and parts of the Conservation Zone – experienced deforestation on about 15% of its previously forested area (corresponding to 956 ha). However, an additional 19% of the rainforest here, which corresponds to about 1216 ha, experienced substantial degradation. Multiple consecutive disturbances affected only small areas within all management zones studied. Overall, about 5400 ha of forest in the four management zones experienced some form of disturbance since 2006.

A different picture appears in the two buffer areas surrounding the mining lease area and Ankerana. Both buffer areas are dominated by mosaic landscapes that are mostly inhabited and managed by local communities practising a mix of permanent agriculture and shifting cultivation. Accordingly, multiple consecutive forest disturbances are higher in the buffer areas, with shares of 5.4% (365 ha) around the lease area and 22% (2511 ha) around Ankerana. Furthermore, the two buffer areas experienced much higher shares of forest degradation, at 35% (2360 ha) and 51% (5794 ha), respectively, than the management zones they surround. On the other hand, deforestation is lower in the mining lease area buffer (6.7%; 451 ha) than in the mining lease area itself (15.5%; 956 ha). However, the opposite is the case in the Ankerana buffer area, where deforestation is higher (4.9%; 553 ha) than within Ankerana (2.4%; 164 ha).

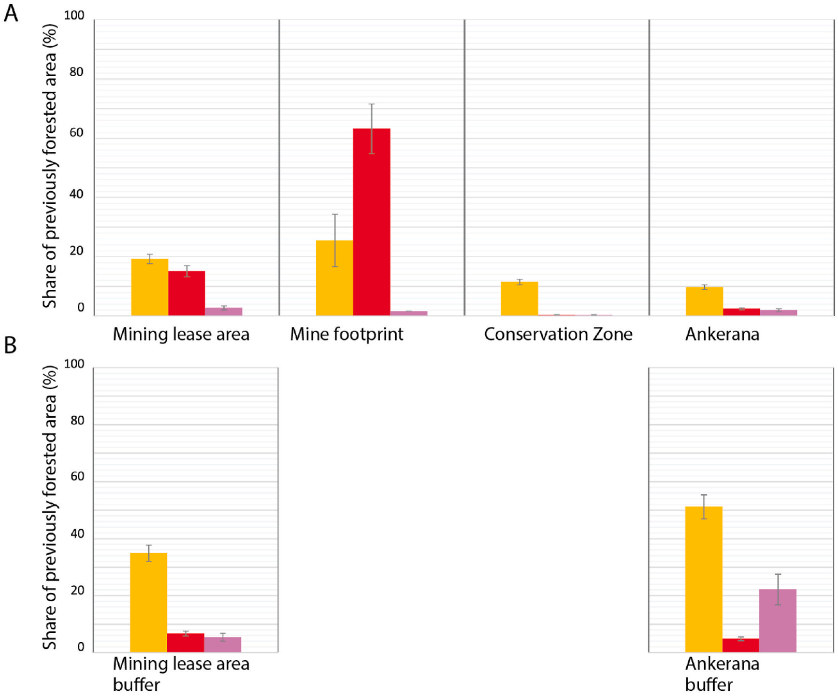

Fig. 4. Shares in per cent of previously forested area that experienced disturbance, by disturbance stratum. Yellow, degradation; red, deforestation; pink, multiple consecutive disturbances. Results are shown for the management zones (A) as well as the buffer areas around the mining lease and Ankerana (B). Important note: the mining lease area includes the mine footprint, parts of the Conservation Zone, and some unprotected land. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Table 1
Areal change observed between 2006 and 2020 (and corresponding standard error) by disturbance stratum, as percentages of previously forested area and in hectares, for the management zones as well as their surrounding buffer areas.

	Management zone							
	Mining lease area		Mine footprint		Conservation Zone		Ankerana	
	%	ha	%	ha	%	ha	%	ha
Deforestation	15.11 ± 1.90	956.35 ± 120.07	63.19 ± 8.36	920.37	0.38 ± 0.05	17.87 ± 2.24	2.41 ± 0.30	164.28 ± 20.63
Degradation	19.22 ± 1.59	1216.38 ± 100.58	25.47 ± 8.83	371.02	11.44 ± 0.95	544.68 ± 45.04	9.73 ± 0.80	662.11 ± 8.27
Multiple consecutive disturbances	2.70 ± 0.69	170.74 ± 43.88	1.64 ± 0	23.89	0.33 ± 0.08	15.66 ± 3.84	1.98 ± 0.48	135.03 ± 32.98
Total	37.03 ± 4.18	2343.47 ± 264.54	90.3 ± 17.19	1315.28	12.15 ± 1.07	578.21 ± 51.12	17.08 ± 1.84	1162.74 ± 125.00
Corresponding buffer area								
Deforestation	6.65 ± 0.84	450.72 ± 56.59					4.88 ± 0.61	553.35 ± 69.47
Degradation	34.84 ± 2.88	2359.75 ± 195.13					51.13 ± 4.23	5793.56 ± 490.08
Multiple consecutive disturbances	5.39 ± 1.36	365.00 ± 92.90					22.16 ± 5.41	2510.80 ± 612.81
Total	46.88 ± 5.08	3175.47 ± 344.62					78.17 ± 10.25	8857.71 ± 1172.36

4.2. Annual deforestation and forest degradation estimates

CODED captures the year of a first disturbance event, making it possible to spatially visualize the year of deforestation or degradation. Looking at the entire study period, most of the deforestation observed occurred within the area of the mine footprint. The earliest deforestation events occurred mostly between 2007 and 2011 – that is, during the construction and implementation of the mine; this was followed by a second deforestation phase between 2013 and 2018 (Figs. 5A and 6A). Outside the mining lease area, mainly in the southern part of the buffer area, relatively large areas were deforested after 2012, with the highest deforestation rates observed from 2013 to 2015 and in 2017, followed by a decrease and then another peak in 2020 (Fig. 7A). South of the mining lease area, CODED also detected many instances of multiple consecutive disturbances (Fig. 5A). These are typically observed in areas with shifting cultivation. Overall, little deforestation was observed in the Conservation Zone next to the mine. In Ankerana, most deforestation events detected (Fig. 5B) occurred between 2008 and 2010, with only few additional patches deforested at a later stage, in 2020 (Fig. 6A), mainly in the eastern and northeastern parts of the biodiversity offset area. In its northeastern corner, these areas of more recent deforestation are interspersed with areas that experienced multiple consecutive disturbances, indicating shifting cultivation activi-

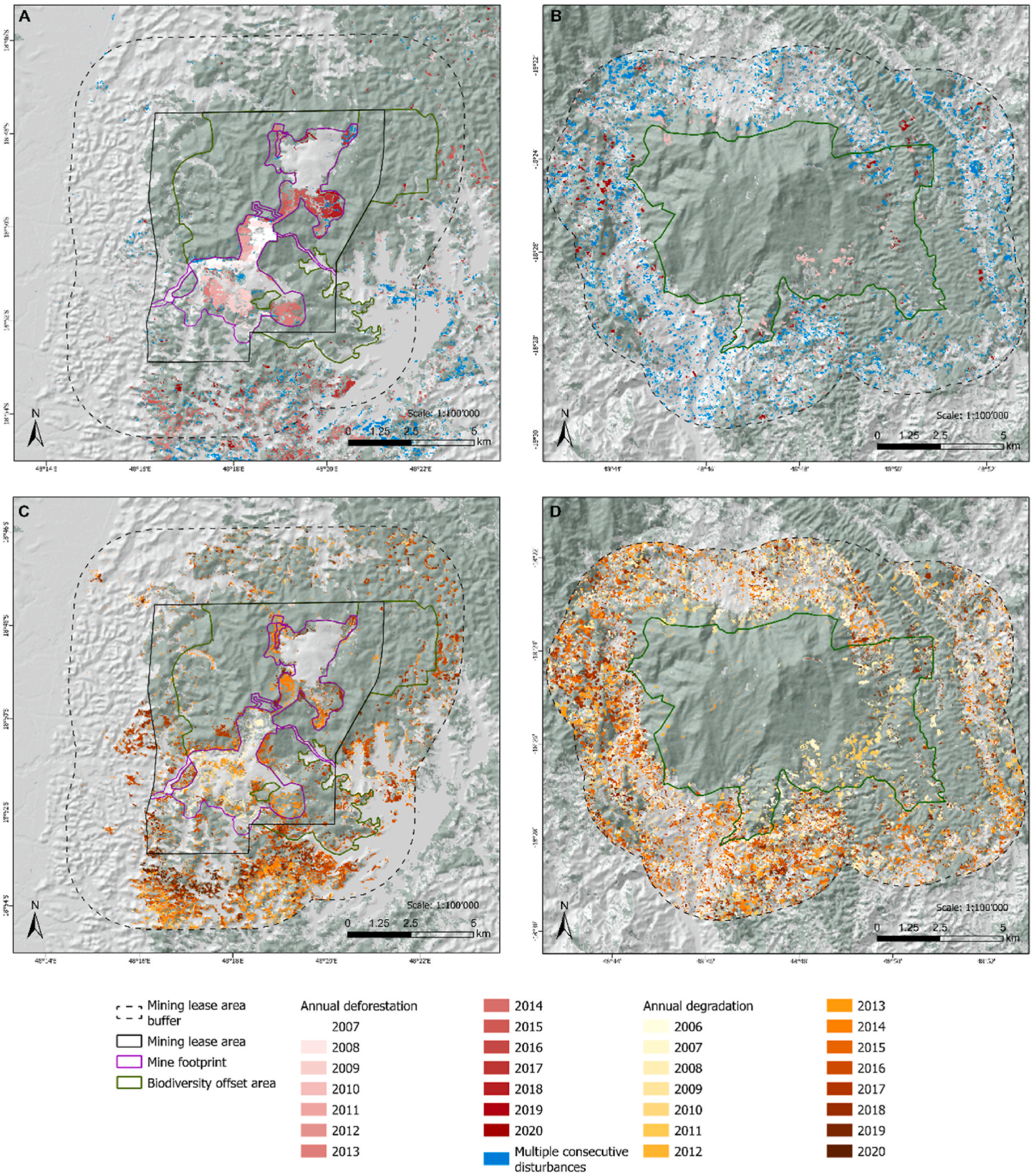

Fig. 5. Annual forest disturbance maps of the analysed management zones and their respective buffer areas for the period of 2006–2020. (A) Annual deforestation in and around the mining lease area, (B) annual deforestation in and around Ankerana, (C) annual forest degradation in and around the mining lease area, (D) annual forest degradation in and around Ankerana. Note: No deforestation was observed for 2006.

ties. Multiple consecutive disturbances were particularly common in the buffer area around Ankerana. This buffer area also experienced continuous deforestation between 2007 and 2020. While in most years this affected only small areas, we observed two pronounced peaks in 2010 and 2020; in 2010, roughly ten times as much forest was cleared as in the preceding years (Fig. 7A).

Forest degradation, on the other hand, was observed in all management zones (Fig. 5C and D), including the biodiversity offset areas. In the mining lease area, the mine footprint, and the adjacent Conservation Zone, forest degradation was relatively low until 2012, with a massive increase in 2013 and continued high levels up to 2017 (Fig. 6B). While degradation within the mine footprint decreased continuously during this period, it increased in the Conservation Zone, returning to low levels from 2018 onwards. A similar temporal pattern occurs in the mining lease area buffer (Fig. 7B).

In Ankerana, forest degradation shows an increasing trend until 2011 (Fig. 6B). After that, degradation levels dropped and stabilized at low values. This is not the case for the Ankerana buffer area, where forest degradation rates were fairly stable until 2011 and then increased sharply in 2013. From then on, degradation rates fluctuated at a very high level compared to the four management zones, although with a decreasing trend (Fig. 7B).

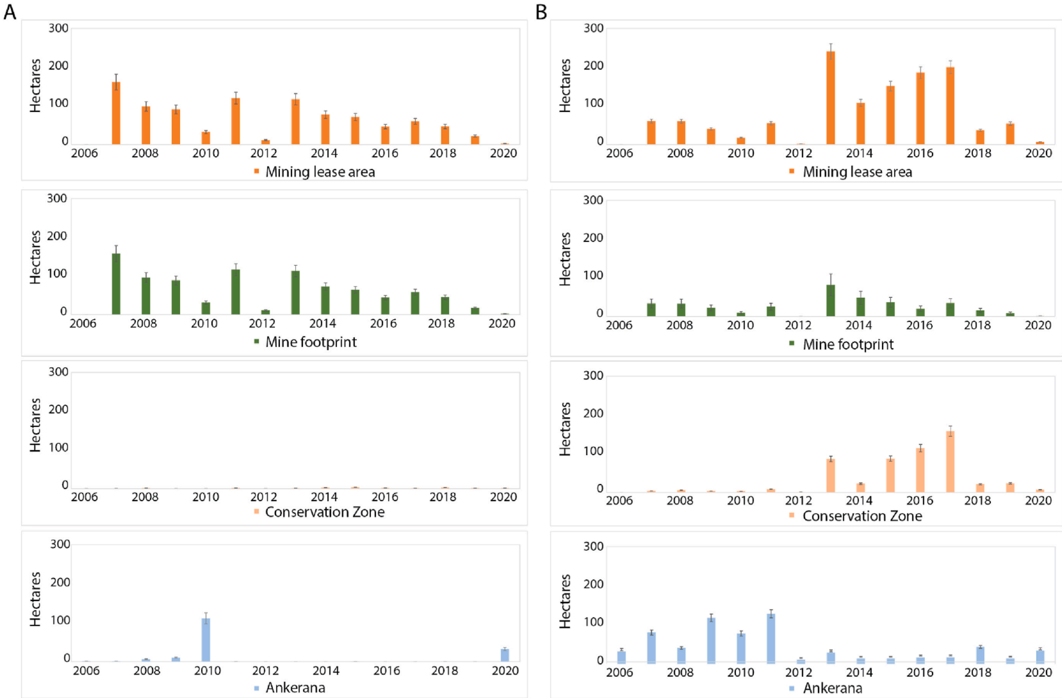

Fig. 6. Annual deforestation (A) and forest degradation (B) in hectares between 2006 and 2020 in the different management zones.

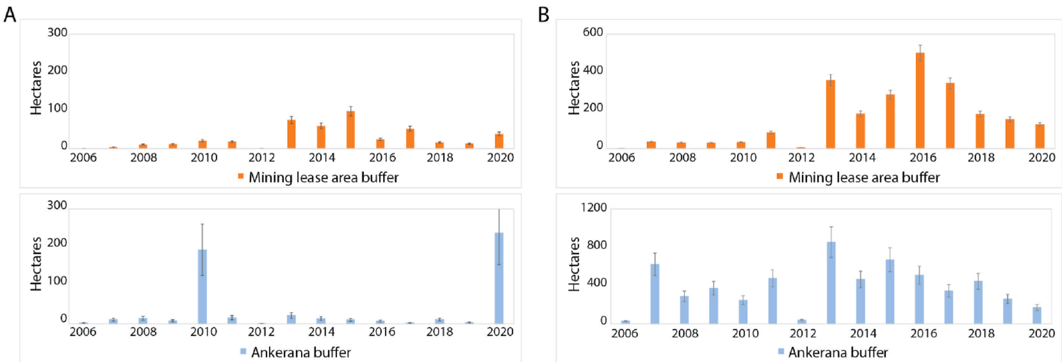

Fig. 7. Annual deforestation (A) and forest degradation (B) between 2006 and 2020 in the buffer areas surrounding the different management zones. Note that the y-axis maximums in (B) are two and four times higher, respectively, than in (A).

4.3. *Classification accuracy*

Table 2 provides the CODED strata classification accuracies representative of the different management zones. Samples (n = 510) were randomly selected, which resulted in some falling into the buffer stratum (n = 41). Samples falling into this stratum reduce the accuracies substantially, even though no "real" misclassification has occurred. Therefore, we provide the accuracy measures in two versions, one considering the buffer stratum, and a second one excluding the buffer stratum.

The accuracies obtained are similar to those found in other studies using CODED (Arévalo et al., 2020a; Aryal et al., 2021; Bullock, 2019; Bullock et al., 2020a, 2020c; Potapov et al., 2017). The accuracy assessment with the buffer stratum shows a marked difference

Table 2
Stratification accuracy estimation for the greater Moramanga area and the study period of 2004–2020. PA, producer's accuracy; UA, user's accuracy; OA, overall accuracy.

	Forest	Non-forest	Deforestation	Degradation	Multiple consecutive disturbances	OA
PA	1.00	0.98	0.52	0.53	0.58	0.88
UA	0.96	0.98	0.87	0.8	0.84	
PA (without buffer stratum)	1.00	0.99	0.66	0.78	0.73	0.88
UA (without buffer stratum)	0.96	0.98	0.87	0.80	0.84	

between user's accuracies (UAs) and producer's accuracies (PAs). UAs are above 80% for all categories. These values indicate a high degree of agreement between the map classification and actual forest development on the ground (assessed by interpreting reference samples) (Olofsson et al., 2014). By contrast, at 52–58%, PAs are fairly low for all disturbance strata. The main reason is misclassifications involving the buffer stratum, which was introduced as a "dummy stratum". PAs increase to 66%–78% when the buffer stratum is excluded from the accuracy assessment. A detailed stratum-specific cross-tabulation matrix can be found in Table S2 in the supplementary data file.

The additional comparison of our deforestation and degradation maps with those of Vancutsem et al. (2021) shows good agreement. For the deforestation classification, we obtained an omission error of 8.5% and a commission error of 3.9%, while for the degradation map we obtained an omission error of 4.5% and a commission error of 10.8%. We did not compare the multiple consecutive disturbances map because there was no comparable class in Vancutsem et al. (2021).

5. Discussion

This study analysed how forest cover change dynamics and rates developed at and around the Ambatovy mine and its main biodiversity offset areas. The aim was to understand how the increasing limitation of local communities' access to forest resources due to the mining lease, the biodiversity offset areas, and other protected areas affects the surrounding mosaic landscape and its precious remaining but increasingly fragmented primary forests – considering not only deforestation, but also forest degradation. Furthermore, we aimed at identifying and separating the forest change (i.e. degradation) pattern caused by shifting cultivation from the remaining degradation captured from space. To date, a number of studies have aimed at developing approaches or ready-to-use datasets providing long-term, highly resolved annual deforestation, forest degradation, and multiple forest disturbance dynamics for larger regions (Bullock et al., 2020a, 2020c; Kennedy et al., 2010; Vancutsem et al., 2021; Zhu and Woodcock, 2014). However, few studies have used these approaches to observe such detailed forest dynamics in relation to the development and operation of large extractive mines in tropical forest landscapes. Xiao et al. (2020) and Liu et al. (2022) used LandTrendr (Kennedy et al., 2012) to monitor on-site mining disturbance and recovery in sparsely vegetated landscapes in China and Mongolia, respectively. Many studies use the Global Forest Change (GFC) datasets (Hansen et al., 2013), which provide global annual updates on per cent forest loss (i.e. deforestation, but not forest degradation or multiple disturbances) per 30 × 30 m Landsat pixel. Devenish et al. (2022) used this dataset as a basis for their study of Ambatovy's efforts to slow deforestation in the defined biodiversity offset areas. Similarly, Mishra et al. (2022b) used the GFC dataset to assess annual deforestation, but not degradation or multiple disturbances, between 2001 and 2019 in Odisha, India. Caballero Espejo et al. (2018) used the same GFC dataset, but in combination with CLASlite (Asner, 2009), to assess both deforestation and forest degradation caused by artisanal gold mining in the Madre de Dios region in Peru. However, their focus was mainly on the direct impacts of mining on tropical forests; they did not assess indirect land use changes. Ultimately, it is the combination of recent developments in technology and algorithms, including access to the entire historical Landsat satellite data archive through cloud computing services, that has enabled us to monitor different types of forest disturbance in such great spatial and temporal detail. This, in turn, makes it possible to better understand how the development of large-scale mining activities impacts the surrounding landscapes.

As expected, the greatest share of deforestation occurred within the mine footprint, with the highest rates observed mostly between 2007 and 2011, which is when the mine was built. After that, deforestation and degradation continued at low levels as the open-pit mining area was gradually expanded. One year after the mine became operational, in 2013, a spike in deforestation rates, and even more so in forest degradation rates, was observed outside the mining lease area, particularly in the unprotected parts of the forest to the south as well as to the east, near Mantadia National Park, the Torotorofotsy Ramsar Wetland, and the Conservation Zone next to the mine. This spatiotemporal pattern suggests a displacement effect (i.e. spillover or leakage) resulting from the reduced availability of forest resources (e.g. charcoal, firewood, construction wood) and land resources for agricultural production. In addition to that, local communities in villages around the mining lease observed increased in-migration to the area starting with the beginning of the construction of the mine (Zaehringer et al., 2024). On the eastern escarpment of Madagascar, forest land is under customary management. When parts of the forest that local people have designated for future agricultural use is "locked away" for conservation purposes, this can force households – especially the more destitute ones – to deforest land managed by customary authorities (even though it formally belongs to the state). This leads to the disappearance of remaining accessible small forest patches in the surroundings of the mining lease area and the various types of protected areas, such as the biodiversity offset areas, national parks, and special reserves. Such a development was also reported by interviewees of Bidaud et al. (2017) and Ramialison and Andriamiadanomenjanahary (2021), and was observed in the context of protected area establishment in similar landscapes (Llopis et al., 2019; Zaehringer et al., 2015).

In both biodiversity offset areas studied – the Conservation Zone next to the mine and the Ankerana Forest Massif – deforestation rates remained low throughout the observation period. This was not the case for forest degradation. We found that substantial areas had become degraded. In Ankerana, degradation occurred particularly between 2006 and 2012. Given that Ankerana became a biodiversity offset area in 2011, the slowing of degradation after 2012 indicates a positive conservation effect. Degradation was also particularly pronounced between 2013 and 2017 in the Conservation Zone, for which we found no reasonable explanation. The deforestation pattern differs between Ankerana and the Conservation Zone, with two deforestation peaks observed both within and around Ankerana in 2010 and 2020, whereas in the area surrounding the Conservation Zone (which corresponds to the mining lease area buffer) deforestation was low until 2013, then increased sharply and remained high until 2015, after which it slowly decreased, indicating a spillover effect, as mentioned above. The amount of degradation in the buffer areas around the mining lease and Ankerana is particularly striking. The temporal pattern is similar, but the rates in the surroundings of Ankerana are much higher compared to the buffer around the mining lease. The amount of degradation can only be explained by a combination of all known factors, including shifting cultivation, selective logging, charcoal and timber production, and firewood collection, even though shifting cultivation should be captured in the "multiple consecutive disturbances" class (see Fig. 5). But this is not surprising, given the moderate accuracy obtained for this stratum. We did not rigorously examine the causal relations between political events, governance processes, and our findings. However, based on information from the scientific literature, conservation news portals, personal conversations with Malagasy scientists, and a household survey with local communities living close to the mining lease (Zaehringer et al., 2024), we believe this pattern could be interpreted as follows. From 2009 to 2013, Madagascar experienced major political turmoil following a coup d'état. During this period, state control over natural resources was substantially weakened. Illicit timber (rosewood) extraction boomed (Randriamalala and Liu, 2010), and shifting cultivation for subsistence rice production was informally expanded at the expense of forests (Llopis et al., 2019; Vieilledent et al., 2018). The second peak in 2020 might be an indirect consequence of the Covid-19 pandemic, which caused general economic pressure as well as reverse migration of the urban poor to their rural villages of origin (Boillat and Zaehringer, 2020; Golar et al., 2020), increasing illegal logging and pressure on forests for agricultural production (Golar et al., 2020; Piquer-Rodríguez et al., 2023). At the same time, funding for protected area management dropped, partly due to the cessation of tourist visits (Piquer-Rodríguez et al., 2023), and deforestation for shifting cultivation peaked in many protected areas on the island, as indicated by massive increases in fires recorded (Eklund et al., 2022). In future similar studies, combining a household survey and a spatial analysis and better aligning the former with the study areas and objectives of the latter could help to better capture the direct and indirect drivers of activities increasing or decreasing deforestation and degradation.

The annual degradation pattern further indicates that the use restrictions introduced in Ankerana in 2012 reduced forest degradation within the biodiversity offset area to almost zero, but may have spurred forest degradation in the surrounding area, with a sharp increase observed in 2013. This pattern indicates a spillover effect similar to that observed in the surroundings of the mining lease area with regard to both deforestation and forest degradation. This is confirmed by the household survey findings of Zaehringer et al. (2024) in which local communities living near the mining lease reported reduced access to forests and forest resources that were now protected or "owned" by a private company. It is also confirmed by the results of Bidaud et al. (2017), who conducted household surveys and interviews in an affected community near Ankerana, where farmers reported that they had experienced increased pressure on land and forest resources in the region since the new restrictions on forest access had come into force. This development was compounded by population growth, as interviewed villagers in the Ankerana region themselves observed (Bidaud et al., 2017). After peaking in 2013, forest degradation rates in Ankerana slowly declined, but remained at very high levels throughout the study period.

Overall, in line with Devenish et al. (2022), our results confirm that conservation interventions through protected areas around the Ambatovy mine effectively prevent deforestation. Furthermore, we show that they also reduce forest degradation within the biodiversity offset areas. In addition, however, we provide new spatiotemporal evidence that in doing so they lead to increased pressure on land and natural resources – and thus increased forest disturbance – outside the boundaries of protected areas (Andam et al., 2008; Bidaud et al., 2017, 2018; Llopis et al., 2019; Ramialison and Andriamiadanomenjanahary, 2021). Other indirect effects of the mine that further heighten the pressure on land around the mining lease area are labour in-migration and urbanization (Jütersonke and Dönges, 2015; Ramialison and Andriamiadanomenjanahary, 2021), as well as expropriation of land and resettlement (Ramialison and Andriamiadanomenjanahary, 2021), which force farmers to change their agricultural practices at the new location or to find new ways of generating income.

Importantly, the observed total area affected by forest degradation is significantly larger than the area affected by deforestation, a development that has also been observed in other tropical regions of the world (Matricardi et al., 2020). In our study, this clearly shows both in the two biodiversity offset areas analysed and – even more so – in the buffer areas surrounding them, indicating that degradation is more widespread and appears to be much more difficult to control than deforestation (Lapola et al., 2023; Mertz et al., 2012). The difficulty of containing forest degradation at a predefined spatial level represents a major threat to sensitive tropical ecosystems (Baccini et al., 2017; Gibson et al., 2011; Vancutsem et al., 2021). In their biodiversity loss offsetting strategy (Berner et al., 2009; Devenish et al., 2022; Hase et al., 2014), Ambatovy projected zero hectares of forest disturbance for the mine-adjacent Conservation Zone, a goal they met only with regard to deforestation, but not with regard to forest degradation. Their biodiversity management strategy focused mostly on preventing direct deforestation, while widely ignoring indirect deforestation as well as forest degradation – two separate and potentially equal or even greater threats to tropical forest ecosystems. In addition, Ambatovy committed themselves to restoring the mining site (Hase et al., 2014). One additional offsetting option would have been to develop, together with local communities, an active restoration programme (e.g. planting of native trees, closing small perforations) for fragmented forest patches (Chen et al., 2023; Hariharan and Raman, 2022) around the mining lease and biodiversity offset areas that could have benefitted local communities (Wainaina et al., 2021). We acknowledge, however, that restoration in the tropics is challenging (Crouzeilles et al., 2017). Our results suggest that taking into account the observed forest degradation would likely even result in a net loss outcome of Ambatovy's biodiversity loss offsetting projects. Generally, it appears that the role of forest degradation as a driver of carbon emissions and biodiversity loss is still largely ignored or neglected, despite the fact that it substantially reduces ecosystem health, ecosystem resilience, and ecosystem service functioning and supply (Barlow et al., 2016; Lapola et al., 2023; Lewis, 2009).

Even though Ambatovy has exclusive management and control rights within the mining lease area, as set out in the Mine Forest Management Plan (Berner et al., 2009), it appears that the company has underestimated the importance of forest degradation in several of the biodiversity offset areas it manages. According to Elias et al. (2018) and based on our own results obtained from a household survey conducted around the mining lease in parallel to this study (Zaehringer et al., 2024), degradation in the Conservation Zone can partly be attributed to unauthorized land use activities of local communities, such as small-scale selective logging and firewood collection. Logging of high-grade timber is particularly difficult to curb, as revenues from the illicit trade in these timbers have increased greatly since the early 2000s (Combaz, 2020). Both the illicit extraction of high-value timber for international trade and the illegal collection of wood for domestic use increased pronouncedly after the 2009 Malagasy coup d'état and remained widespread during the subsequent years of political transition. The freezing of most international aid following the crisis, combined with a severe reduction in national revenues, led the public authorities to seek new sources of funding. These were found in land leases to large companies active in the mining, agriculture, and tourism sectors, or in the trade in illegally logged rosewood, the export of which was legalized by government decree (Combaz, 2020; Randriamalala and Liu, 2010; Razafindrakoto et al., 2017). In the absence of any comprehensive control mechanism, trafficking of natural resources established itself as a permanent and integral part of Madagascar's political economy while exerting unprecedented pressure on the country's forest ecosystems (Allnutt et al., 2013; Combaz, 2020; Jones et al., 2019). All these aspects are likely to have contributed, to varying degrees, to the continued decline in forest cover and widespread forest degradation observed in this study. It should be noted, however, that Madagascar's Mining Code has undergone various revisions since 2020. Major changes approved include an increase in royalties due to the state and to local authorities, a reduction of the maximum area of land granted to permit holders, the protection of sites of high religious or cultural significance, stricter environmental standards, and mandatory social accountability for mine operators (Adewole, 2023). However, Madagascar's legal and policy framework for environmental and social impact assessments continues to have various limitations that should be addressed (Nikièma et al., 2023).(

Ambatovy's activities may also have indirect and off-site impacts that contribute to forest degradation. Some of these may be unintentional. Such impacts remain largely unconsidered in mining companies' compensation plans, and in most cases, communities living near the mining assets are left to bear the burden of the environmental damage caused (Antwi et al., 2017; Ballet et al., 2019; Huff, 2016; Mabey et al., 2020; Seagle, 2012). A number of such unexpected side effects have also been caused by Ambatovy's activities. For example, the slurry pipeline and tailings dams have caused soil erosion along the pipeline (Berner et al., 2009; Dynatec Corporation of Canada, 2006a), destruction of rice fields (Ambatovy, 2015), as well as water and soil pollution (e.g. tailings dam leaks, sulphur dioxide spills) in their surroundings and further downstream (Ballet et al., 2019; Soustras, 2017; Soustras and Randrianarisoa, 2018). Other reported damage to the environment and people's well-being includes spraying of insecticides (Soustras, 2017), unintentional introduction of invasive species (Moore et al., 2015), and air pollution (Soustras, 2017; Soustras and Randrianarisoa, 2018).

5.1. Considerations regarding satellite data and methodology

The generally low deforestation and degradation values detected for 2012 are likely due to the lack in available satellite data for this specific year (see Fig. S1 in the supplementary data file). Also, based on the algorithm design, disturbances at the beginning and end of the observation period (2004–2006 and 2019–2020) cannot be estimated or may be underestimated because there are no or not enough pre- and post-disturbance satellite observations (Bullock, 2019). For example, areas classified as deforestation in 2019 and 2020 that are used for shifting cultivation might move to the multiple disturbances stratum in the near future. Furthermore, if the disturbance occurred early in the study period, it is likely that not only degraded, but also some deforested areas have since regenerated into secondary forests. Finally, areas that were not forested in 2004 and developed into secondary forest without experiencing any disturbance were not specifically classified into one of the disturbance strata. However, comprehensive visual inspection of high-resolution satellite imagery in Google Earth Pro (2021) overlaid with the forest cover mask computed for the year 2020 suggests that such cases are rare in the areas analysed.

Unbiased area estimates and confidence intervals were calculated based on the validation data collected in our specific study region. They are thus valid for this particular region, but not necessarily for others. Applying the methodology to another region in a different agroecological and socio-economic context will require recalculating the estimates – a process that can be time-consuming and labour-intensive.

Likely reasons for a lower PA for the degradation stratum (52%) include the fact that degradation is not represented as a complete clearing of the canopy but rather a status somewhere in between. In addition to that, several degradation effects, for example vertical structural changes such as a decrease in foliar density or plant area density, might not be visible from above. The associated spectral change might often be too small compared to natural variability to trigger a change detection in CODED, resulting in limited detection and an underestimation of forest degradation that has actually occurred (Bullock, 2019). Another explanation is that many degradation pixels from the sample interpretation were removed from the final classification because they fell into the buffer stratum, which confirms that many of these pixels are located near disturbance events and are likely to have experienced some disturbance. Taking this into account when calculating accuracies would increase PA for the degradation stratum to 78%.

The low PA for the deforestation stratum is due to confusion with the forest degradation stratum. According to Bullock (2019), CODED correctly identifies disturbances but then erroneously classifies them as deforestation instead of degradation. Furthermore, CODED has been shown to be more prone to omission errors than commission errors when detecting disturbance pixels (Bullock et al., 2020c). This is particularly the case when the disturbance stratum is small in area compared to one of the stable strata (Arévalo et al., 2020b; Bullock, 2019; Bullock et al., 2020a, 2020c), a situation which at the same time also results in an underestimation of the unbiased areas obtained from the sample of a particular disturbance stratum. This, in turn, reduces the impact of the omission errors on the variance estimate for that disturbance stratum. Furthermore, by requiring multiple consecutive observations to exceed a statistical boundary, the change detection approach is robust to false detection of change (Bullock et al., 2020a). To account for forest disturbance patterns typical of Madagascar (i.e. shifting cultivation), we modified CODED and introduced strata representing multiple disturbance patterns, such as (rotational) regrowth and disturbance, or degradation and subsequent deforestation. But due to high classification errors among these three strata we decided to merge them into one stratum representing *multiple forest disturbances*.

Another important point is that the two buffer areas around the mining lease and Ankerana are artificial zones delimited solely for the purpose of our analysis. The aim was to detect deforestation and forest degradation spillover effects in the immediate vicinity of the Ambatovy mine and its biodiversity offset areas indicated by differences in the observed deforestation and forest degradation rates in the surroundings of the management zones compared to within them, on the one hand, and by a distinct, corresponding temporal deforestation or degradation trend pattern within and around the mining lease and the Ankerana biodiversity offset area. The absolute numbers and percent shares of the entire buffer area are again representative only of this artificial zone and are therefore of limited informational value. However, they are relevant for the mine owners insofar as they indicate that the total biodiversity offsetting capacity of the action taken to compensate for forest loss due to the establishment of the mine is reduced due to the high rates of forest degradation observed in the surroundings of the biodiversity offset areas.

6. Conclusion

In this study, we assessed forest change dynamics and differentiated various types of forest disturbance within and around the Ambatovy mining lease area and the Ankerana Forest Massif – Ambatovy's main biodiversity offset area – between 2006 and 2020. To do so, we successfully adapted and applied the CODED algorithm, implementing it in GEE and obtaining reliable accuracies and areal estimates. While CODED is a useful and promising tool to differentiate deforestation and forest degradation, our adapted version is not yet capable of reliably capturing consecutive or rotational forest cover changes indicating shifting cultivation. Furthermore, it is not capable of providing information about the level of relative forest biomass loss caused by degradation, which could give an indication of the intensity of forest use, including shifting cultivation, and support the development of suitable policy responses.

Based on the detailed spatiotemporal deforestation and forest degradation pattern we obtained, we conclude that the mine and its direct and indirect effects have increased pressure on land and forest resources in the surroundings of the mining lease area and the company's main biodiversity offset area. The fact that deforestation and forest degradation in the unprotected areas surrounding the mining lease area – including the Conservation Zone – and the Ankerana Forest Massif increased after the implementation of the mine and the biodiversity offset areas indicates spillover effects. Such spillover effects may cause substantial additional adverse environmental and social impacts in the vicinity of biodiversity offset areas, which should be considered in future biodiversity offsetting policies and best practices.

The pervasive effects of widespread forest degradation may magnify biodiversity loss well beyond the level prevented by halting deforestation in the biodiversity offset areas (Barlow et al., 2016). Forest degradation has largely been ignored in international and national policies, including biodiversity offsetting policies (Matricardi et al., 2020), one reason being that forest degradation is challenging to assess and monitor. Remote sensing research has been addressing this challenge in recent years, producing datasets and algorithms that make it possible to monitor forest degradation, thereby contributing to a better understanding of related biodiversity loss and providing a basis for accountability reporting (Bullock et al., 2020a; Ferraz et al., 2003; Matricardi et al., 2020; Vancutsem et al., 2021). In the case of Ambatovy, if forest degradation were taken into account in their biodiversity offset programme, this could well result in a net biodiversity loss outcome. Accordingly, it is crucial that biodiversity offsetting policy mechanisms become more holistic, by extending their outcome assessments beyond protected offset areas, and more inclusive, by considering potential social impacts on local communities in addition to biodiversity outcomes.

CRediT authorship contribution statement

Sandra Eckert: Writing – review & editing, Writing – original draft, Validation, Supervision, Conceptualization. **Luc Schmid:** Writing – review & editing, Writing – original draft, Validation, Software, Formal analysis, Data curation. **Peter Messerli:** Supervision. **Julie G. Zaehringer:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.rsase.2024.101269.

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