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1 **Accelerating global mountain forest loss threatens biodiversity hotspots**

2
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16

17 **SUMMARY**

18 The frontier of forest loss has encroached into mountains in some regions. However, the global
19 distribution of forest loss in mountain areas, which are home to >85% of the world's birds,
20 mammals, and amphibians, is uncertain. Here we combine multiple datasets, including global
21 forest change and selected species distributions, to examine spatiotemporal patterns, drivers
22 and impacts of mountain forest loss. We find 78.1 Mha of montane forest was lost during 2001–
23 2018 and annual loss accelerated significantly, with recent losses being 2.7-fold greater than
24 those at the beginning of the century. Key drivers of mountain forest loss include commercial
25 forestry, agriculture, and wildfire. Areas with the greatest forest loss overlap with important
26 tropical biodiversity hotspots. Our results indicate protected areas within mountain biodiversity
27 hotspots experienced lower loss rates than their surroundings. Increasing the area of protection
28 in mountains should be central to preserving montane forests and biodiversity in the future.

29

30

31 **INTRODUCTION**

32 Mountains are vital to the world’s terrestrial biodiversity as they provide habitat to more than
33 85% of the world’s bird, mammal, and amphibian species¹. Montane forests serve as important
34 refuges for large numbers of rare and endangered species with small geographical distributions,
35 making them represent regions of high conservation significance². As many montane species
36 have narrow ranges³, even relatively small reductions in forest habitat may increase their risk
37 of extinction. Unfortunately, forest loss and degradation pose significant threats to the
38 persistence of forest dwelling species that rely on specific microenvironments worldwide⁴. In
39 addition, climate change is forcing many montane species to move to higher elevations in
40 search of suitable habitats^{5,6}, but their ability to do so is potentially limited by topographic
41 constraints and the integrity of the habitat⁷. Understanding the dynamics of mountain forest
42 loss worldwide is therefore crucial for predicting and mitigating the potential impacts on
43 sensitive forest species⁸.

44

45 Mountain forest loss was historically limited in many areas as high elevations and steep slopes
46 presented physical barriers to human exploitation⁹. As such, most forest exploitation occurred
47 in more accessible lowland areas for a variety of activities including logging and agriculture¹⁰⁻
48 ¹². However, since the turn of the 21st century, mountain forests have been increasingly
49 exploited for timber and wood products, as well as to support emerging agricultural systems,
50 such as boom crops and tree-based plantations, for example in Southeast Asia¹³⁻¹⁵. These
51 activities have reshaped montane forests, potentially reducing the size and number of refuge
52 areas, increasing the risk of extinction of forest dwelling species¹⁶, and weakening the ability
53 of forests to store carbon¹³ and regulate climate¹⁷. Elsewhere, such as in the Andes, there is
54 reported evidence of an overall net gain in woody vegetation, the dynamics of which vary with
55 elevation¹⁸. There, mountain forest losses dominated vegetation change at lower elevations
56 (1,000–1,499 m) from 2011 to 2014, but forest gains occurred at higher elevations above 1,500
57 m¹⁸. Regional reports¹³⁻¹⁸ that are often based on a diverse array of locally derived data and
58 varying analytical approaches, may not necessarily contribute to the determination of clear and
59 generalized trends in mountain forest loss at a global scale, leading to difficulties in assessing
60 the impact of forest loss over mountain regions. Thus, a wider global analysis—with a common
61 analytical framework—conducted in the 21st century when there is evidence of the frontier of
62 forest loss encroaching into mountains, is required to accurately understand mountain forest
63 loss patterns, trends, drivers and impacts worldwide. This information is essential for
64 developing effective biodiversity conservation and forest management strategies in the future.

65

66 Here, we conducted a comprehensive assessment of global mountain forest loss during the first
67 two decades of the 21st century. We first assessed forest loss patterns across global mountains
68 and determined the proportion of areas showing signs of regrowth. Second, we determined the
69 extent of mountain forest loss within biodiversity hotspots across a range of elevation gradients,
70 as elevation regulates biophysical climate impacts¹⁷ and therefore potentially reshapes
71 expected species responses to climate change¹⁹. Third, we estimated the fraction of mountain
72 forest loss within mountain biodiversity hotspots in and around protected areas (PAs). We also
73 examined the drivers of mountain forest loss by comparing our mountain forest loss maps and
74 statistics with other recently developed land-use maps²⁰. We find that annual forest loss
75 accelerated significantly across global mountains during the first two decades of the 21st
76 century. Unfortunately, many of areas with the greatest mountain forest loss overlap with
77 critical tropical biodiversity hotspots. Forestry caused the greatest mountain forest loss at the
78 global scale. However, within biodiversity hotspots, commodity agriculture was the main
79 driver of mountain forest loss in Southeast Asia and shifting cultivation was preeminent in
80 tropical Africa and South America. Our results also emphasize the significance of protected
81 areas in conserving forest-dependent biodiversity in mountains and provide a strong foundation
82 for creating region-specific conservation recommendations aimed at preserving forests and the
83 biodiversity they harbour.

84

85 **RESULTS**

86 *Patterns and drivers of mountain forest change*

87 Mountain forests covered 1,100 million hectares (Mha) globally in 2000 (Table 1).
88 Approximately 78.1 Mha of forest loss occurred in mountain regions between 2001 and 2018,
89 which constitutes a relative gross loss of 7.1% worldwide since 2000 (Table S1). Mean annual
90 gross loss was 4.3 Mha yr⁻¹, equivalent to 0.39% yr⁻¹ (Table 1). We found that mountain forest
91 loss was significantly accelerating worldwide, with a rate at 0.202 Mha yr⁻² ($p < 0.01$).
92 Importantly, there was a striking difference in mountain forest loss rate between periods before
93 and after 2010. Annual forest loss in mountains increased more than 1.5 fold from <3.5 Mha
94 yr⁻¹ during 2001 to 2009 to 5.2 Mha yr⁻¹ during the period 2010 to 2018. Tropical mountains
95 experienced the most rapid acceleration, with the annual loss after 2010 being twice that before
96 2010. This transition was probably related to the rapid expansion of agriculture into highland
97 areas, for example in mainland Southeast Asia^{14,15}, as well as increased exploitation of
98 mountain forest products as lowland forests became depleted or were the focus of greater forest

99 protection.

100

101 Between 2001 and 2018, global mountain forest loss reached a prominent peak in 2016 (about
102 65% higher than in the previous year). This surge was mainly driven by forest loss in Asian
103 mountains (Fig. 1A). Compared with the 2016 peak, annual mountain forest loss decreased in
104 2017 and 2018, but the annual loss in these two years (mean of 6.5 Mha yr⁻¹) remained high
105 compared with the earlier years of the 21st century. The key activities associated with mountain
106 forest loss were commercial forestry (42%), followed by wildfires (29%), shifting cultivation
107 (15%) and commodity agriculture (10%; Fig. 3A). These drivers starkly contrast with the
108 activities reported recently for global forest loss²⁰. While our focus was forest loss, we note
109 that substantial gains in mountain forests have also occurred worldwide. Using a sample-based
110 method^{21,22}, we found that 23.14% (1,157 of 5,000 pixels) of the forest loss areas at some point
111 during 2001–2018 experienced some degree of tree cover regrowth by 2019 (Fig. S1;
112 Supplementary Data S1). For the whole period 2000–2018, the annual net rate of mountain
113 forest loss, accounting for both forest losses and gains, was 0.31% per year (Table 1).

114

115 Five of seven global regions (Asia, South America, Africa, Europe, and Australia) experienced
116 significant acceleration in mountain forest loss during the period of observation, with North
117 America and Oceania being exceptions (Fig. 1A; Table 1). Over the 18-year study period, the
118 greatest loss of mountain forest area occurred in Asia (39.8 Mha), accounting for more than
119 half of the global total (Table 1). This increase in mountain forest loss primarily occurred in
120 southern Asia ($\leq 30^\circ\text{N}$), where high population densities potentially have a negative effect on
121 forest cover and integrity^{23,24}. However, the trend in mountain forest loss in northern Asia was
122 not significant (Table 1). We also find clear regional differences in the drivers of mountain
123 forest loss and the proportion of forest gain within Asia (Tables 1 and S1). Mountain forest loss
124 in northern Asia ($>30^\circ\text{N}$) was primarily attributed to wildfire (e.g., Russia); and this region
125 experienced only a small proportion of forest gain ($\sim 15\%$). By contrast, mountain forest loss
126 in southern Asia was driven by commercial forestry (e.g., in southern China) and commodity
127 agriculture (e.g., in Indonesia, Vietnam and Myanmar); and $\sim 40\%$ of loss areas showed signs
128 of regrowth—in part, due to the maturation of plantation trees (Table S1; Fig. S1). North
129 America had the second greatest mountain forest loss area (18.7 Mha; 24% of global mountain
130 forest loss), with $\sim 16\%$ of forest gain (Table 1). This proportional gain was less than half that
131 in South America ($\sim 33\%$) and thus the annual net rate of forest loss in North America (0.41%

132 yr⁻¹) was more than twice that of South America (0.19% yr⁻¹; Table 1). Africa experienced the
133 greatest relative forest loss of 0.54% yr⁻¹ and had the smallest proportional forest gain of 15.4%.
134 Therefore, the annual net rate of mountain forest loss in Africa was greater than that of any
135 other region at 0.48% per year (Table 1).

136

137 Globally, substantial mountain forest losses occurred at elevations <1,000 m, where >70% of
138 forest gain also occurred (Fig. S2). From the 2000s to 2010s, there was a large increase in forest
139 loss at low-to-moderate elevations, particularly below 1,000 m (Fig. S3). This pattern of
140 increased forest loss at low elevations might obscure the fact that forest loss is creeping
141 upwards. Further, temporal patterns indicate increases in forest loss at higher elevations in Asia,
142 South America, and particularly Africa (Fig. S4B,D,E). In Asia, the peak of forest loss in 2016
143 was primarily concentrated at 100–300 m, but extended up to 1,200 m, which largely followed
144 the global pattern (Fig. S4A–B). In North America, most mountain forest loss was concentrated
145 in 2004 and 2005 at elevations below 1,000 m (Fig. S4C). In South America, Africa, and Europe,
146 mountain forest loss reached a peak in 2017 at elevations of about 250 m, 300 m, and 500 m
147 elevation, respectively (Fig. S4D–F). In contrast, mountain forest loss in Australia did not
148 follow a particular trend with respect to elevation, but there were specific years (in 2003, 2007,
149 2009, 2013, and 2016) with significant loss (Fig. S4G) that were linked to drought and
150 bushfires²⁵⁻²⁸.

151

152 We found significant increases in mountain forest loss in tropical and temperate latitudes, but
153 not at boreal latitudes (Fig. 1B). Tropical montane forests, which experienced the greatest loss
154 (32.9 Mha; 42% of global mountain forest loss), also had the fastest acceleration of loss at
155 0.131 Mha yr⁻² (Fig. 1B; Table 1). Around 31.2% of these losses have shown signs of regrowth,
156 which is higher than that of temperate and boreal regions (Table 1). Our results show that the
157 dominant drivers of mountain forest loss in the tropics were shifting cultivation (44%),
158 commodity agriculture (28%), and commercial forestry (24%; Fig. 3A). In Indonesia, the
159 tropical country with the greatest loss of mountain forests at 3.97 Mha (relative loss of 7.1%),
160 commodity agriculture was the dominant driver (Table S1). Forest loss in Laos (3.08 Mha;
161 16.4%) and Vietnam (2.81 Mha; 17.8%) was also substantial (Table S1). Parts of Laos, Vietnam
162 and northern Thailand (1.29 Mha; 7.9%) form a cluster in mainland Southeast Asia where
163 agriculture-driven deforestation has moved to higher elevations in recent decades^{15,29}. The loss
164 of forest in Myanmar (2.80 Mha; 8.8%), which was affected by both commercial forestry and
165 commodity agriculture (Table S1), was likely related to its recent re-engagement with regional

166 and global economies³⁰. Malaysia was ranked number 10 worldwide in mountain forest loss
167 (2.2 Mha; 16.4%) (Table S1), with the most loss occurring in Peninsular Malaysia, where oil
168 palm expansion before 2010 was an important driver (Fig. 2A)³¹. These Southeast Asian
169 countries were all also in the top 10 with respect to acceleration in mountain forest loss (Table
170 S1; Fig. 2B). In those regions, the loss was primarily attributed to deforestation in mountains
171 through permanent land-use change for commodity production (Table S1), for example, rubber,
172 oil palm, and feed corn^{20,32}; this process can also be validated by sample-based manual
173 interpretation (Supplementary Data). Brazil has experienced well-publicized lowland forest
174 loss in recent decades³³. Our results show that Brazil also experienced 2.26 Mha (7.6%) of
175 mountain forest loss driven largely by shifting cultivation (Table S1). This result highlights the
176 different drivers of mountain versus lowland forest loss, for which the latter is widely reported
177 to be caused by conversion for commodity agriculture (e.g., soy)³⁴ and grazing³⁵. Also
178 associated with shifting cultivation is the loss of montane forests in other South American
179 countries (e.g., Colombia and Peru) and in Africa (e.g., Guinea and Madagascar), with a total
180 loss of 4.99 Mha in these four countries (Table S1).

181

182 Temperate montane forests had the second greatest area of losses between 2001 and 2018 (27.9
183 Mha; 36% of the global total). The primary cause of these losses was commercial forestry, with
184 more than 75% of the area lost being attributed to this sector (Fig. 3A). Despite the large area
185 lost, temperate montane forests had the smallest annual decrease among all the forests studied,
186 with a rate of 0.28% per year (Table 1). In the mountains of the United States, forest loss in the
187 west was greater than in the east (Fig. 2A); the leading cause was commercial forestry, followed
188 by wildfire (Table S1). Most mountain forest loss in temperate China occurred in the southern
189 mountains with a fast pace of loss (Fig. 2B) and was primarily driven by commercial forestry
190 (Table S1). Elsewhere, absolute losses of mountain forests were small in Europe, but countries
191 like Portugal, Ireland, and the United Kingdom had substantial percentage losses relative to
192 forest cover in 2000. Again, commercial forestry contributed to >90% of losses in these
193 countries (Table S1).

194

195 Losses in boreal regions were comparatively small than at lower latitudes, but in some years
196 montane forest losses at these high latitude locations rivalled those found in temperate areas,
197 and were on the order of 1.6 to 2.1 Mha yr⁻¹ (Fig. 1B). The rate of acceleration in losses of
198 boreal mountain forests was also very low (0.016 Mha yr⁻²; Table 1). Russia and Canada
199 experienced a large amount of mountain forest loss: 11.95 Mha (6.9%) and 5.57 Mha (7.4%),

200 respectively (Table S1). Wildfire (69%) was the dominant disturbance to boreal montane
201 forests (Fig. 3A). However, the lack of a significant trend in boreal mountain forest loss (Fig.
202 1B; Table 1) may suggest that the reported increase in boreal wildfires³⁶ only affects montane
203 forests in particular years, and does not constitute a long-term threat. Mountain forest gain in
204 boreal regions was the smallest observed (12.5%; Table 1). The annual net rate of forest loss
205 was therefore greater than in tropical and temperate regions, at 0.39% per year (Table 1).

206

207 As tree plantations have expanded greatly worldwide over the last few decades³⁷, their removal
208 contributes to forest loss rates reported here. To test what proportion of tree plantation removal
209 accounted for mountain forest loss, we separated the forest loss into naturally regenerating
210 forests and plantations using new data on global forest management³⁸. We confirmed that
211 nearly 70% of the global mountain forest loss occurred in naturally regenerating forests (Fig.
212 4). At the regional scale, we showed naturally regenerating forests in the boreal zone accounted
213 for the largest proportion of the loss (74%), while in the tropics, one third of mountain forest
214 loss occurred in plantations (Fig. 4). Crucially we found that the proportion of mountain forest
215 loss occurring in plantations has not changed over the analysis period (Table S2), providing
216 evidence that the expansion of plantation forests does not explain the large acceleration in
217 mountain forest loss reported here. This independent analysis confirms that the majority of
218 mountain forest loss is occurring in natural forests.

219

220 *Forest loss within mountain biodiversity hotspots*

221 To map biodiversity hotspots, we focused on two species pools: one for all species of
222 amphibians, birds, and mammals listed on the International Union for Conservation of Nature
223 (IUCN) Red List and the second for threatened species only. We used two metrics: range-size
224 rarity (RSR) and species richness (SR). RSR, a measure of endemism^{39,40}, is a reliable
225 indicator of mountain biodiversity as endemism is positively associated with elevational
226 ranges⁴¹. SR represents the total number of species present. Our mapping of mountain
227 biodiversity hotspots shows they are primarily concentrated in tropical regions although they
228 vary somewhat by the species pool (all or threatened) and the metric of hotspot definition (RSR
229 versus SR; Figs. S5 & S6). The distribution of RSR hotspots is similar for all species and
230 threatened species, including in Sundaland, Wallacea, the Philippines, Madagascar, western
231 Ecuador, tropical Andes, Brazil's Atlantic forest, and Mesoamerica (Fig. S5). By contrast, SR
232 hotspots vary widely for all and threatened species (Fig. S6). SR hotspots for all species have
233 a small range probably because the most abundant species tend to inhabit the lowlands, not

234 mountain areas, while SR hotspots for threatened species are concentrated in mountainous
235 areas in southwestern China and Southeast Asia that contain the world's largest number of
236 endangered species.

237

238 Total forest loss in mountain biodiversity hotspots over the 18-year study period ranged from
239 1.4 to 14.4 Mha (or 3.8 to 6.2%), depending on the index used. The loss for mountain forests
240 in the hotspots for threatened species was 11.0 to 14.4 Mha (5.5 to 6.2%). Importantly, relative
241 forest loss was greater in mountain hotspots for threatened species than for all species under
242 the same index (Table 2). Further, the acceleration of forest loss in mountain biodiversity
243 hotspots ($0.005\text{--}0.064\text{ Mha yr}^{-2}$) was significant ($p < 0.01$; Table 2) regardless of the species
244 pool and the metric of hotspot definition. RSR hotspots, for which such areas comprise a larger
245 proportion of the global distribution of species³, occur at all elevations from 0 to 3,500 m, with
246 high RSR values located above 2,000 m (Fig. 5). At any elevation, RSR hotspots for threatened
247 species experienced greater relative mountain forest loss than for all species. Mountain forest
248 loss in RSR hotspots reached the peak at about 100 m, then decayed exponentially with
249 increasing elevation, with half occurring at about 350 m (Fig. 5). Although the greatest RSR
250 values were found higher than where most forest loss occurred, substantial forest loss did occur
251 at those elevations (i.e., approximately 2,500-3,000 m) (Fig. 5; Table S3).

252

253 Within RSR hotspots for threatened species, nearly half of forest loss was associated with
254 shifting cultivation (47%); the other two major activities were commodity agriculture (23%)
255 and commercial forestry (23%; Fig. 3B). The six countries with the greatest mountain forest
256 loss within RSR (threatened) hotspots were Indonesia (1.62 Mha), Malaysia (0.95 Mha),
257 Madagascar (0.75 Mha), Vietnam (0.71 Mha), Colombia (0.69 Mha), and Peru (0.62 Mha;
258 Table S4). In the Southeast Asian countries, commodity agriculture was the main driver of
259 mountain forest loss within the hotspots, whereas in tropical Africa and South America, shifting
260 cultivation was preeminent (Tables S3). In terms of relative loss of montane forests in
261 biodiversity hotspots, more than half of the top 10 countries were in Africa: South Africa
262 (27.71%), Zimbabwe (27.64%), Guinea (24.79%), Côte d'Ivoire (22.55%), Madagascar
263 (15.38%), and Mozambique (12.33%); the remaining four were in Chile (34.48%), Mongolia
264 (30.10%), Canada (14.96%), and Malaysia (13.34; Table S4). The four countries with the
265 greatest acceleration in montane forest loss in biodiversity hotspots were Indonesia,
266 Madagascar, Vietnam, and Malaysia, ranging from $\sim 3,200$ to $4,850\text{ ha yr}^{-2}$ (Table S4).

267

268 *Mountain forest loss in protected areas within hotspots*

269 Protected area coverage (proportion of forest area within PAs) is the largest in the SR (all)
270 biodiversity hotspots, with more than half of hotspot areas included within PAs (Table 2). In
271 some cases, this coverage can approach 100% in areas with very high elevations above 3,500
272 m (Fig. S7). In RSR hotspots, only 30% of mountain forest within hotspots was included in
273 PAs (Table 2), suggesting there is a large proportion of forest area with high rates of species
274 endemism that is unprotected. At high elevations (>3,000 m), more than 35% of forest area
275 within RSR hotspots is protected (Fig. 5; Table S3). However, there are some countries with
276 low PA coverage for mountainous forests in biodiversity hotspots, particularly Angola and
277 Papua New Guinea, where PA coverage is <1% (Table S5).

278
279 In all types of mountain biodiversity hotspots, relative forest loss inside PAs was much less
280 than outside (Table 2), suggesting that PAs within mountain biodiversity hotspots may be
281 effective in limiting forest loss (ratio of relative forest loss inside versus outside of PAs less
282 than 1). However, the trends depend somewhat on the metric and pool of species considered.
283 Relative forest loss within RSR hotspots in PAs was lower than outside of PAs at all elevations,
284 albeit less so at high elevations (Fig. 5). In contrast, within SR hotspots, the distribution varied
285 when all versus threatened species are considered. Relative mountain forest loss was less in
286 PAs than outside at elevations up to 3,000 m for all species; but for threatened species, lower
287 loss inside PAs only occurred for the elevation band ranging from 400 to 1,900 m (Fig. S7).

288
289 In the RSR (threatened) hotspots inside PAs the dominant drivers of forest loss were shifting
290 cultivation (38.3%), commodity agriculture (33.1%) and commercial forestry (24.9%; Fig. 3B).
291 The lowest relative forest loss ratio inside versus outside PAs was found in RSR hotspots where
292 commodity agriculture was the dominant driver, while the highest ratio was observed in
293 hotspots where shifting cultivation and commercial forestry were the main drivers (Fig. S8). In
294 most countries, PAs were associated with reduced forest loss relative to their surrounding areas
295 within hotspots (Table S5). For example, Brunei, Chile, Canada, and New Zealand have the
296 lowest ratios of relative forest loss inside versus outside of PAs within hotspots (Table S5).
297 However, in some countries, such as Côte d'Ivoire, Haiti and Nicaragua where shifting
298 cultivation dominates, relative forest loss inside PAs is more than twice that outside (Table S5).
299 The same is true for Russia where wildfire was the main cause of mountain forest loss.

300

301 **DISCUSSION**

302 Our global analysis renders three important findings: (1) Mountain forest loss has accelerated
303 significantly throughout most of the first two decades of the 21st century, encroaching upon
304 areas of known high conservation value to terrestrial biodiversity; (2) Various types of shifting
305 cultivation emerges as the most frequent driver of mountain forest loss in the tropics, but
306 commodity agriculture and forestry activities are also key drivers; (3) Protected areas generally
307 have been effective in curbing mountain forest loss within their boundaries inside biodiversity
308 hotspots, particularly where commodity agriculture is the dominant deforestation driver.
309 However, we find great variation on these three issues throughout the world.

310
311 About three quarters of the 129 countries we analysed experienced an acceleration of mountain
312 forest loss (Table S1). Most of the countries with the greatest acceleration were within
313 Southeast Asia. Parts of India and southern China also experienced substantial losses. These
314 regions with large acceleration align with many of the world's most sensitive biodiversity
315 hotspots for mammals, birds, and amphibians—thus substantial negative impacts to critical
316 habitat have likely already occurred^{42,43}. While we did not yet see a major upward shift in the
317 elevation of forest loss at the global scale of analysis, this transition has been reported before
318 for Southeast Asia¹³. Further, the history of the progression of forest loss in mountain areas
319 suggests such a shift will likely unfold in locations with high forest pressure but limited
320 capacity to protect forest lands from location-specific drivers, mostly related to agriculture
321 expansion, forest product acquisition, and logging (including illegal). Increased encroachment
322 resulting in forest loss into these sensitive areas directly increases the risk of species extinction
323 and/or forces other species to migrate upward if possible.

324
325 Agricultural expansion is of concern worldwide with respect to forest loss⁴⁴. The greatest
326 acceleration of mountain forest loss occurred in countries where shifting cultivation or
327 commodity agriculture were dominant (Table S6), highlighting the importance of agricultural
328 expansion in mountain regions. Encroachment of shifting cultivation in highland forests is
329 problematic to address in countries where this form of agriculture contributes to food and
330 livelihood security of rural communities^{45,46} and where intensification of cultivation can lead
331 to negative consequences for biodiversity and climate^{47,48}. Forest lands are often viewed as an
332 ownerless public resource and are therefore utilized as needed by individuals for food and
333 livelihood security⁴⁹. A complicating issue is that contemporary protected area boundaries are
334 often established in areas where people have lived and exploited the forest long before
335 governments recognized the need to conserve and manage them, with varied impacts on human

336 welfare⁵⁰. In cases where profit-driven commodity agriculture is the driver of mountain forest
337 loss, intervention can be effective when the will to enforce forest protection laws is strong. An
338 example is found on the border areas of Thailand and Laos where maize cultivation on forested
339 lands is being phased out by the Thai government, but in Laos the exploitation of forest for
340 lucrative boom crops persists^{51,52} (Fig. S9). This situation demonstrates the drastic outcome in
341 forest loss patterns related to differing institutional efficiency and capacity to enforce existing
342 forest conservation policies. Further, the economic situation in Laos and its geographical
343 location in Southeast Asia make it susceptible to external investments that drive deforestation
344 for agriculture, timber/wood products, and energy⁵³.

345

346 We recognize the importance of promoting the regeneration of converted forests both naturally
347 and through forestation programs. While we find that much regrowth has occurred in the
348 locations of mountain forest loss worldwide, two issues are critical within the scope of our
349 analysis. Firstly, reforestation with native species is preferable over the establishment of mono-
350 specific tree plantations, which by some definitions are considered a type of forest. Secondly,
351 initial disturbance causing forest loss may critically damage the habitat of sensitive species to
352 the extent that they may not recover when forests reappear. Another issue is that the wellbeing
353 of other types of organisms that contribute to biodiversity should be considered. Regarding
354 sensitive species in biodiversity hotspots, the critical issue extends beyond simply preventing
355 forest loss to also maintaining the integrity of forests in large enough zones to allow natural
356 movements and sufficient space for ranging species. Protected areas should be designed with
357 this purpose in mind.

358

359 Regionally distinctive drivers of mountain forest loss mean that efforts to curb the acceleration
360 of mountain forest loss will require regionally appropriate interventions. In regions where
361 shifting cultivation is a strong driver, like in Brazil, Colombia and Peru, efforts should be made
362 to ensure agriculture does not impact frontier (intact or primary) forests where possible. Rather,
363 it would be better to preserve biodiversity to establish new agriculture ventures where forests
364 are already disturbed or land has been recently cleared⁵⁴. Whereas in regions where commodity
365 production is more prevalent (e.g., Indonesia, Vietnam, and Malaysia), increased commitment
366 is needed urgently to halt commodity-driven forest loss and safeguard mountain forest
367 biodiversity. Given that human population pressure has also been a major cause of biodiversity
368 loss in PAs in the past few decades⁵⁵, we recommend that relevant strategies should consider

369 balancing economic development, biodiversity conservation and sustainable livelihoods
370 especially within and surrounding PAs.

371

372 We see examples where the existence of PAs has significantly reduced forest loss, compared
373 with the areas surrounding them. Recent studies have also demonstrated the role of PAs
374 worldwide in preventing forest loss^{56,57}. Largely in agreement, we find that of the 78 countries
375 with data pertaining to PAs in montane areas, about half were effective in keeping forest loss
376 to be less than half of the loss experienced outside of PAs (Table S5). Unfortunately, in 12
377 countries the forest loss inside the PAs was greater than or equivalent to that outside. Drivers
378 of mountain forest loss inside PAs tend to vary, with shifting cultivation, commercial forestry
379 and commodity agriculture being important in a variety of locations. The strategic expansion
380 and development of new PAs are thus promising avenues to improving mountain forest
381 conservation for biodiversity now and into the future, especially in countries where PA
382 coverage is low^{58,59}. Many countries have only marginally effective PAs because, even in areas
383 where forests are protected, there are destructive anthropogenic activities (e.g., logging) taking
384 place that tax sensitive organisms. In these places, there is ample opportunity for improved PA
385 management, and more adequate resourcing, and stricter enforcement of laws and regulations
386 designed to protect forests.

387

388 As alluded to above, any new measures to protect mountain forests should be adapted to local
389 conditions and contexts⁶⁰, and they should reconcile the need for enhanced forest protection
390 with ensuring food production and human wellbeing⁶¹. More integrated socio-ecological
391 research is needed to improve our understanding of biodiversity and ecosystem functioning in
392 complex and sensitive mountain ecosystems, especially at the interface between social and
393 natural systems. Such knowledge should boost awareness of the importance of preserving
394 forest integrity whilst maintaining or enhancing human wellbeing, and, hopefully, help change
395 attitudes regarding the reliance on destructive food production and energy generation systems.

396

397 In closing, our global analysis of mountain forest loss identifies an alarming acceleration in
398 mountain forest lost worldwide over the last two decades. Important drivers have been various
399 types of agriculture, forestry, and wildfire, with regional differences. These global results
400 provide a foundation for further regional and local studies to examine nuanced differences more
401 closely. Our analysis also highlights the importance of appropriately managed protected areas
402 in preserving mountain forest biodiversity in the face of increasing human pressures for food

403 production and a changing climate. By providing a clear understanding of the current trends
404 and drivers of mountain forest loss, we hope to this analysis will inform and support
405 conservation efforts aimed at preserving critical montane forest ecosystems for future
406 generations.

407

408 **EXPERIMENTAL PROCEDURES**

409 *Resource Availability*

410 *Lead Contact*

411 Requests for further information should be directed to and will be fulfilled by the lead contact,
412 Zhenzhong Zeng (zengzz@sustech.edu.cn).

413 *Materials Availability*

414 This study did not generate new unique materials.

415 *Data and Code Availability*

416 The original data generated during this study are available at Mendeley Data,
417 <https://data.mendeley.com/datasets/myym96xcdy/1> and
418 <https://data.mendeley.com/datasets/t67hc9k7gd/1>. Code used to analyse and plot data have
419 been deposited at https://github.com/hexinyue33/mountain_forests. Any additional
420 information required to reanalyse the data reported in this paper is available from the lead
421 contact upon request.

422

423 *Data sources*

424 *Global forest change data and visual interpretation for forest gain*

425 We used a Global Forest Change (GFC) dataset from 2000 to 2018 (version 1.6, available at
426 https://earthenginepartners.appspot.com/science-2013-global-forest/download_v1.6.html)⁶² to
427 analyse forest loss over mountains during the 21st century. This dataset uses Landsat satellite
428 images to detect annual forest cover loss at a 1 arc-second resolution (~30 meters at the equator),
429 spanning latitudes from 80°N–50°S. The global dataset is divided into 10° × 10° tiles (each
430 containing 40,000 by 40,000 pixels). Trees are defined as “all vegetation taller than 5 m in
431 height”⁶². Forest loss is “stand-replacement disturbance”⁶², which includes both permanent loss
432 (conversion to another land use) and temporary loss (e.g., loss from a forest fire). We first
433 created a baseline forest cover map in 2000 from the percent tree cover layer using the criteria
434 of Hansen et al.⁶³ that forest cover comprises at least 25% tree canopy cover at the pixel scale
435 (30 × 30 m), which is an appropriate threshold for multispectral imagery to unambiguously
436 identify tall woody vegetation. To investigate the degree to which our results were sensitive to
437 the choice of threshold, we also used a tree-cover threshold of 50% to define forests for
438 comparison (Fig. S10). Then, we mapped forest loss for all years in the 2001 to 2018 period
439 from the forest loss layer at the pixel level. Forest loss area is the sum of all pixel areas where
440 forest loss occurred. To distinguish the change of pixels with latitude, we calculated the pixel
441 area as a function of latitude: pixel area = cos(latitude) × pixel area at the equator.

442

443 To check whether there was subsequent regrowth around 2019 where the forest was lost during
444 the study period 2001-2018, we performed an independent assessment of forest gain using a
445 sample-based approach following recommendations from Global Forest Watch²¹ and good
446 practice guidance of Olofsson et al.²². We randomly sampled 5,000 pixels that experienced
447 forest loss (Supplementary Data) using random number generation, and visually interpreted
448 forest gain using very-high-resolution imagery from Google Earth and Planet Explorer. We
449 started with Google Earth for visual interpretation because it has a very high resolution (ranging
450 from 15 m to even 15 cm); if there was no clear satellite image in 2019, we expanded the time
451 range to the two years before and after, i.e., 2017-2020, but the image is at least a year after
452 forest loss occurred. For the remaining points that have no images in Google Earth, we changed
453 to Planet Explorer at a resolution of ~3.7 m for interpretation using daily or monthly imagery.

454

455 *Drivers of forest loss*

456 We determined drivers of forest loss using the dataset generated by Curtis et al.²⁰. This dataset
457 shows the dominant driver of forest loss at each 10 km grid cell. There are five categories of
458 drivers of forest loss, including commodity-driven deforestation which is defined as permanent
459 and/or long-term clearing of trees to other land uses (e.g., commodity agriculture), shifting
460 cultivation, forestry (a combination of logging, plantations and other forestry operations with
461 visible forest regrowth in subsequent years), wildfire, and urbanization. The grids that were
462 marked as zero or minor loss in the driver dataset are categorized as “other”. We resampled
463 data from 10 km resolution to 30 m using the nearest neighbour method, to match the scale of
464 global forest cover change data. We then reported the proportion of each driver of mountain
465 forest loss for each country.

466

467 *Topography data*

468 A digital elevation model and global mountain polygons were applied to quantify the
469 topographic pattern of forest loss. We used a high-resolution (30-m) elevation dataset from the
470 Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) Global Digital
471 Elevation Model (GDEM, version 3, available at <https://earthdata.nasa.gov/>)⁶⁴ to quantify the
472 elevational gradients of mountain forest loss. The ASTER GDEM was generated by stacking
473 the observed cloud-masked and non-cloud-masked scene DEMs, spanning latitudes from 83°N
474 – 83°S⁶⁵⁻⁶⁷. Each tile of data has a dimension of 3,601 × 3,601 pixels, or a 1° latitude by 1°
475 longitude area²³. As the tile size of ASTER GDEM differs slightly from that of the forest change

476 data, we first resampled each 1° x 1° DEM tile to 4,000 × 4,000 by using the cubic convolution
477 method and then merged it into a tile of 10° latitude by 10° longitude pixels as in the forest
478 change dataset (i.e., 40,000 × 40,000 pixels).

479
480 We used the Global Mountain Biodiversity Assessment (GMBA) definition (version 1.2,
481 available at www.mountainbiodiversity.org,
482 https://ilias.unibe.ch/goto_ilias3_unibe_cat_1000515.html)⁶⁸ to identify mountain regions,
483 which adopts a ruggedness threshold indicating that the geometrical slope between the lowest
484 and the highest point within a 2.5' pixel must exceed 200 m⁶⁹. The GMBA mountain definitions
485 have the advantage of excluding some unstructured terrain such as large plateaus and expansive
486 valleys or basins, while also not limiting mountains to particular elevations. Based on this
487 definition, the world's mountainous terrain occupies about 1.64 billion ha and accounts for
488 12.3% of the total land area. It uses the GMBA definition along with expert delineations to
489 provide a worldwide inventory of 1048 distinct mountain systems as vector polygons.
490 Mountain regions are divided into eight mega-regions (mostly continents): Asia, Africa, Europe,
491 Australia, North America, South America, Oceania, and Greenland⁶⁸. Although mountain areas
492 in Greenland occupy 4.3% of the total land area in the region, these mountains contain no tree
493 cover and so are not considered here. In the analysis, we also examined forest loss in tropical
494 (24°S to 24°N), boreal (≥50°N) and temperate (residual) regions.

495
496 *Biodiversity hotspots*

497 We identified biodiversity hotspots for amphibians, birds, and mammals (as they have been the
498 most comprehensively assessed and thus polygon maps are available) based on two species
499 pools: (1) all assessed species belonging to any International Union for Conservation of Nature
500 (IUCN) Red List category; and (2) threatened species listed as CR (Critically Endangered), EN
501 (Endangered) and VU (Vulnerable) on the IUCN Red List. Thus, the second pool is a subset of
502 the first. Note that the dataset used a filtering process that eliminates records of Extinct (EX)
503 and Extinct in the wild (EW) from the start. For each of the two species pools, we used existing
504 maps of range-size rarity (RSR) and species richness (SR) based on the raw IUCN ranges
505 (available at <https://www.iucnredlist.org/resources/other-spatial-downloads>). RSR within each
506 pixel is calculated as the pixel area divided by the total distribution area of each species that
507 occurs within this pixel and then summed across all these species to determine the aggregate
508 importance of each pixel. SR represents the total number of species potentially occurring in
509 each pixel (including the possibility of presence and the uncertainty of seasonal occurrence of

510 a species). We therefore used four raster layers consisting of all combinations of the two
511 biodiversity indicators (RSR and SR) and the two species pools (all and threatened). The
512 resolution of these rasters is about 5 km at the equator, but we resampled them to ~30 m to
513 match global forest change data for calculation in our analysis.

514
515 In this dataset, RSR values range from 0 to ~0.72 (for all species) and from 0 to ~0.29 (for
516 threatened species); SR values range from 1 to 912 (for all species) and from 1 to 59 (for
517 threatened species). For each raster, we defined biodiversity hotspots as the upper 2.5% of land
518 cells with the highest RSR or SR values as done previously⁷⁰ and clipped it to the boundaries
519 of the mountain range delineations. The four biodiversity hotspot criteria are referred to as: (1)
520 RSR (all); (2) RSR (threatened); (3) SR (all); and (4) SR (threatened). In each type of
521 biodiversity hotspot within the mountain extent, RSR values range from 0.00073 to ~0.19 (for
522 all species) and from 0.00012 to ~0.29 (for threatened species); SR values range from 675 to
523 847 (for all species), and from 24 to 59 (for threatened species) respectively; these ranges were
524 calculated based on the upper 2.5% the land area.

525

526 *Protected areas*

527 To investigate how much of the area of forest loss within biodiversity hotspots has been
528 protected, we used polygon delineations of PAs from the World Database on Protected Areas
529 (WDPA; available at <https://www.protectedplanet.net/en/thematic-areas/wdpa?tab=WDPA>).
530 We did not include PAs represented by points, as forest loss calculations required areas. Due to
531 the large size of the database, the data were divided into three shapefile layers. We clipped
532 these layers to the extents of our mountain range boundaries separately and then merged them
533 into one layer. A total of 30,515 PA polygons within the mountain range delineations was
534 obtained. All pre-processing was performed in ArcMap 10.6.

535

536 *Data analysis*

537 We assessed temporal, spatial, and elevational patterns of forest loss across global mountains
538 and within mountain biodiversity hotspots. We estimated annual forest loss area occurring in
539 years between 2001 and 2018, beginning from the reference year 2000. Relative forest loss is
540 based on forest cover in the baseline year 2000, calculated as the amount of forest lost in the
541 region relative to the amount of forest that was there (relative forest loss = forest loss area/forest
542 cover in 2000), providing information about rates of forest loss. We evaluated the temporal
543 trend in annual forest loss (i.e., acceleration) using a non-parametric Theil-Sen estimator

544 regression method⁷¹ due to its robustness for trend detection and insensitivity to outliers, which
545 has been widely used in previous research, including in forest cover trend analysis^{72,73}. We then
546 assessed the significance of the trends using the Mann-Kendall (MK) test⁷⁴. To make our results
547 more comparable among different regions or climate zones, we used a standardized annual rate
548 of forest loss proposed by Puyravaud⁷⁵, calculated as: $r = (1/(t_2-t_1)) \times \ln(A_2/A_1)$ where A_1 and
549 A_2 are the forest cover at time t_1 and t_2 . In our analysis, A_1 is forest cover in the baseline year
550 2000 (obtained by the existing tree cover layer as mentioned above) and A_2 is forest cover in
551 2018 (= forest cover in 2000 – forest loss 2001 to 2018 + forest gain).

552

553 To visualize mountain forest loss area occurring at different elevations, we grouped elevation
554 into 50 m bins within 0.5° grid cells. In mountain biodiversity hotspots, we calculated mean
555 RSR (and overall SR) patterns within each elevation bin to represent the potential impacts of
556 elevation-specific forest loss on biodiversity. We then compared the amount of forest loss in
557 mountain hotspots of all species with those associated with threatened species to reveal the
558 differences between various species pools affected by mountain forest loss. Finally, we
559 specifically calculated each country's mountain forest loss for the RSR biodiversity hotspot
560 with threatened species.

561

562 To assess the elevation-specific patterns of PA protection, we calculated PA coverage (i.e.,
563 fraction of forest in PAs) as the ratio of mountain forest within PAs in hotspots versus mountain
564 forest in the corresponding hotspots. We also compare mountain forest loss within biodiversity
565 hotspots inside PAs and outside of PAs at different elevations. In this study, we use the ratio of
566 relative mountain forest loss within biodiversity hotspots inside versus outside of PAs to assess
567 forest loss in the context of PAs (i.e., when the ratio <1, PAs experienced less forest loss than
568 unprotected areas).

569

570 *Uncertainties and limitations*

571 The GFC product we used, does not distinguish between natural forests and tree plantations^{76,77}.
572 Forest loss estimates therefore include forestry activities within tree plantations. Another
573 difficulty we encountered was distinguishing forest (tree) loss from selective logging, which
574 tends to degrade forests rather than resulting in a transition to another type of land cover. Not
575 only permanent forest loss poses direct threats to montane forest biodiversity, but other forms
576 of temporary loss (including partial tree removal) and forest degradation at large spatial scales
577 are threatening to biodiversity, particularly in sensitive habitats like cloud forests, wetlands

578 mountain patches in valleys, etc. Although forestry is an important driver of mountain forest
579 loss as we reported, our independent analysis of forest loss and plantation loss confirms that
580 the majority of loss occurs in natural forests, with less than 20% occurring in plantations (Fig.
581 4). Thus, the forest loss estimates presented in this study are likely to be conservative.

582

583 We acknowledge that our results are based on vertebrate (amphibians, birds, and mammals)
584 distributions only, and that a more thorough investigation of the impacts of forest loss on other
585 taxonomic groups such as plants, fungi, protists, and other types of wildlife (e.g., fish, insects)
586 is needed. As the realm of most organisms (e.g., freshwater protists, fungi and other soil
587 community members including bacteria, protozoa, nematodes, arthropods) is largely unknown,
588 potentially important services offered by entire mountain forest ecosystems may soon be lost,
589 or at least degraded following forest removal⁷⁸.

590

591 Finally, some geographic mountainous areas of known forest loss were not detected in our
592 analysis (e.g., the islands of Timor-Leste and Dominica⁷⁹). The reason for the omission of these
593 countries, and possibly others, is the definition of mountains following the GMBA definition⁶⁸.
594 Although regrettable, as this paper is a global analysis, we used a standard global definition of
595 mountains.

596

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609

610 **Author contributions**

611 Conceptualization, X.H., Z.Z., D.V.S. and J.H.; methodology, X.H.; investigation, X.H;
612 writing – original draft, X.H.; writing – review & editing, A.D.Z., P.R.E., Y.F., J.C.A.B., S.L.,
613 J.H., D.V.S., and Z.Z.; supervision, Z.Z., D.V.S. and J.H.

614

615 **Declaration of interests**

616 The authors declare no competing interests.

617

618

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879 **Table 1. Mountain forest cover change in different regions/climates (2000 to 2018).** Mountain forests in 2000 is the area of mountain forest
880 based on the tree cover threshold of 25% in the year 2000 (Mha). Total mountain forest loss 2001–2018 is the total loss during the period (Mha).
881 Annual relative forest loss (gross) is the mean of relative forest loss (= mountain forest loss/mountain forest cover in 2000) over the 18 years in
882 the region (%). Mountain forest loss acceleration is the gradient in mountain forest loss with time in the region (Mha yr⁻²), determined from the
883 regression of annual loss (dependent variable, which is a rate in ha yr⁻¹) and year (independent) using Theil-Sen estimator, thus, the units of Mha
884 yr⁻². Mountain forest gain proportion is independently estimated by forest gain divided by the total sample size in the region (%). Annual net rate
885 of mountain forest loss is calculated by a standardized method proposed by Puyravaud⁵⁵, by comparing forest cover in the same region in 2000
886 and 2018 (% per year). Asia was separated into northern and southern Asia, with a boundary of 30°N.

Region	Mountain forest area in 2000 (Mha)	Total mountain forest loss 2001–2018 (Mha)	Annual relative mountain forest loss (%)	Mountain forest loss acceleration (10⁻² Mha yr⁻²)	Mountain forest gain proportion (%)	Annual net rate of mountain forest loss (% per year)
Asia	560.5	39.8	0.39	12.2 (*)	27.0	0.30
Northern Asia	255.8	14.1	0.31	1.0	14.9	0.27
Southern Asia	304.7	25.7	0.47	11.4 (*)	39.9	0.29
North America †	220.5	18.7	0.47	1.5	15.9	0.41
South America	158.9	8.3	0.29	1.4 (*)	33.2	0.19
Africa	66.0	6.4	0.54	2.8 (*)	15.4	0.48
Europe	71.9	3.4	0.26	0.9 (*)	16.4	0.22
Australia	15.0	1.0	0.38	0.2	47.4	0.20
Oceania	7.2	0.4	0.32	0.1 (*)	46.7	0.17
Global	1100.0	78.0	0.39	20.2 (*)	23.2	0.31
Tropical	436.1	32.9	0.42	13.1 (*)	31.2	0.30
Temperate	419.9	27.9	0.37	4.6 (*)	27.3	0.28
Boreal	244.0	17.2	0.39	1.6	12.5	0.35

887 * indicates a significant trend at 95% confidence interval (Mann-Kendall test).

888 [†] North America includes Mexico, central American countries (Belize, Costa Rica, El Salvador, Guatemala, Honduras, Nicaragua, and Panama)
889 and nearby island countries of Cuba, Jamaica, Haiti, Dominican Republic, and Trinidad and Tobago.
890

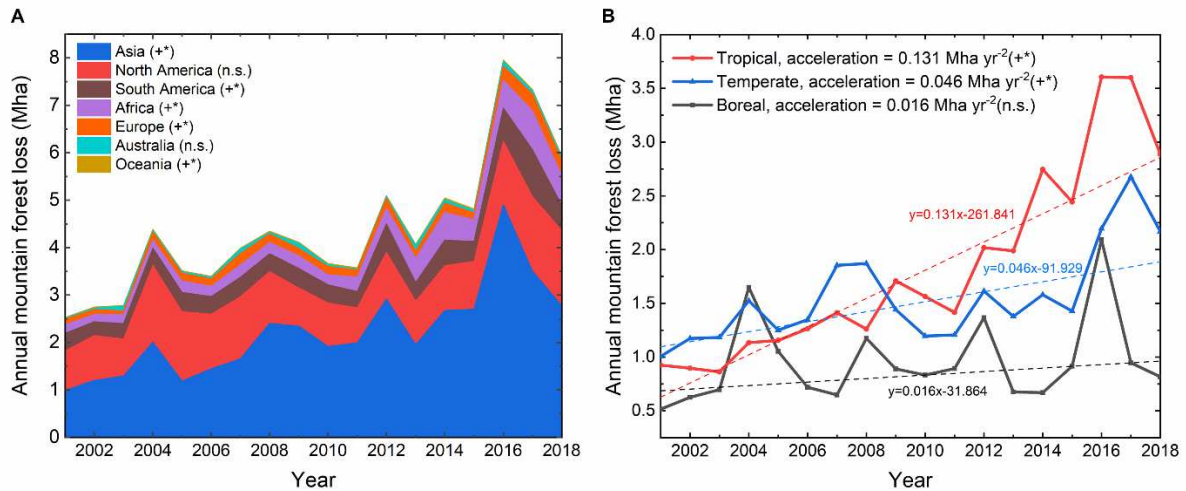
891 **Table 2. Comparison of mountain forest loss within different types of biodiversity hotspots.** Proportion of forest (or loss) within protected
 892 areas (PAs) is the forest (or loss) area within PAs divided by the forest (loss) area in the corresponding hotspots. Relative forest loss inside (or
 893 outside of) PAs is percent forest loss relative to forest cover in the baseline year 2000 inside (or outside of) PAs within hotspots.

Hotspot Types	Forest area in 2000 (Mha)	Total forest loss 2001–2018 (Mha)	Relative forest loss 2001–2018 (%)	Forest loss acceleration (10^{-2} Mha yr⁻²)	Proportion of forest area within PA (%)	Proportion of forest loss within PA (%)	Relative forest loss inside PA (%)	Relative forest loss outside of PA (%)
RSR (all)	223.32	12.98	5.81	4.10 (*)	28.32	15.07	3.09	6.89
RSR (threatened)	177.62	11.03	6.21	3.66 (*)	29.79	16.75	3.49	7.36
SR (all)	37.49	1.43	3.81	0.48 (*)	58.98	21.95	1.42	7.26
SR (threatened)	260.15	14.41	5.54	6.40 (*)	13.14	9.02	3.80	5.80

894 * indicates a significant trend at 95% confidence interval (Mann-Kendall test).

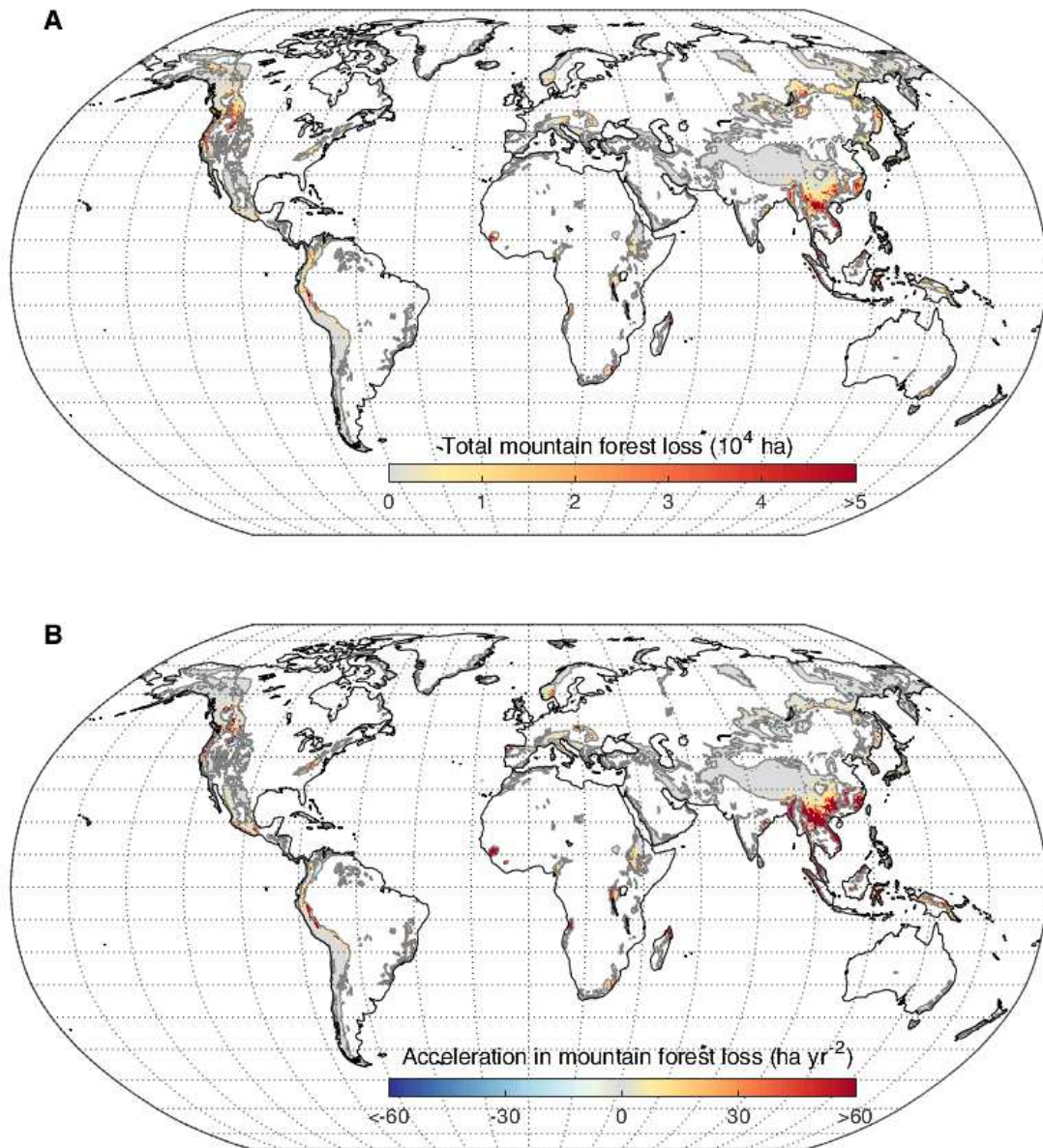
895 RSR: range-size rarity; SR: species richness.

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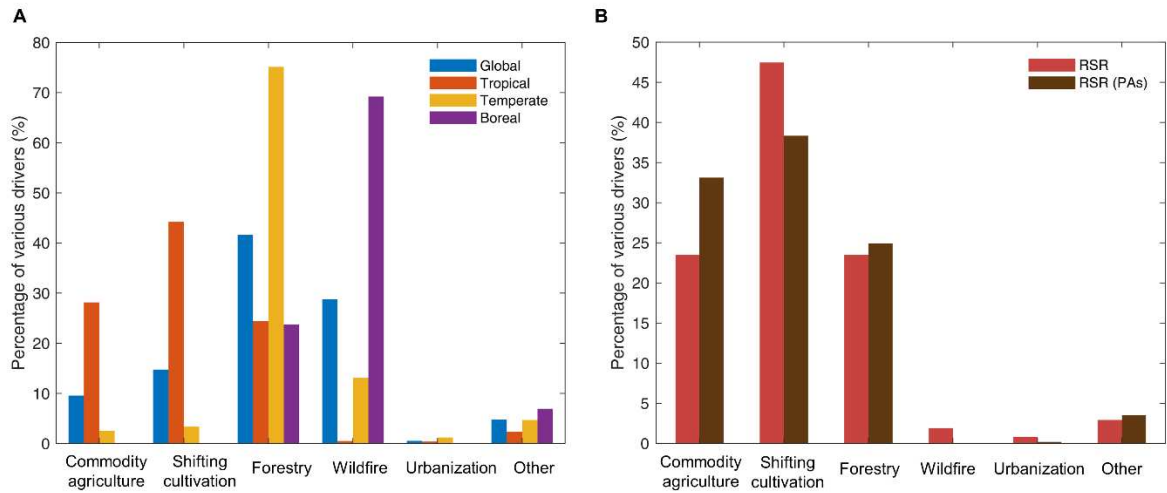
898 **Figure 1. Time series of annual mountain forest loss from 2001 to 2018.** (A) Annual
 899 mountain forest loss in different continents. The total area of all seven regions for each year
 900 represents global mountain forest loss since the baseline year 2000 (i.e., the area is stacked, not
 901 superimposed). A symbol (+*) after the region shows a significant positive trend in mountain
 902 forest loss at the 95% confidence interval; (n.s.) means no significant trend in mountain forest
 903 loss. Trends are determined for the entire 2001–2018 forest loss time series. The loss areas for
 904 Oceania are comparatively small, which appear as a black line. (B) Annual mountain forest
 905 loss in tropical (24°S to 24°N), boreal ($\geq 50^\circ\text{N}$), and temperate (residual) regions. Dashed lines
 906 are trend lines for annual mountain forest loss in tropical (red), temperate (blue), and boreal
 907 (black) regions, estimated by Theil-Sen estimator regression.



908

909 **Figure 2. Spatial pattern of mountain forest loss in the 21st century.** (A) Total mountain
 910 forest loss area. (B) Acceleration in mountain forest loss in 0.5° cells. Mountain regions in grey
 911 show mountains with either little forest loss area or no obvious change during the period.

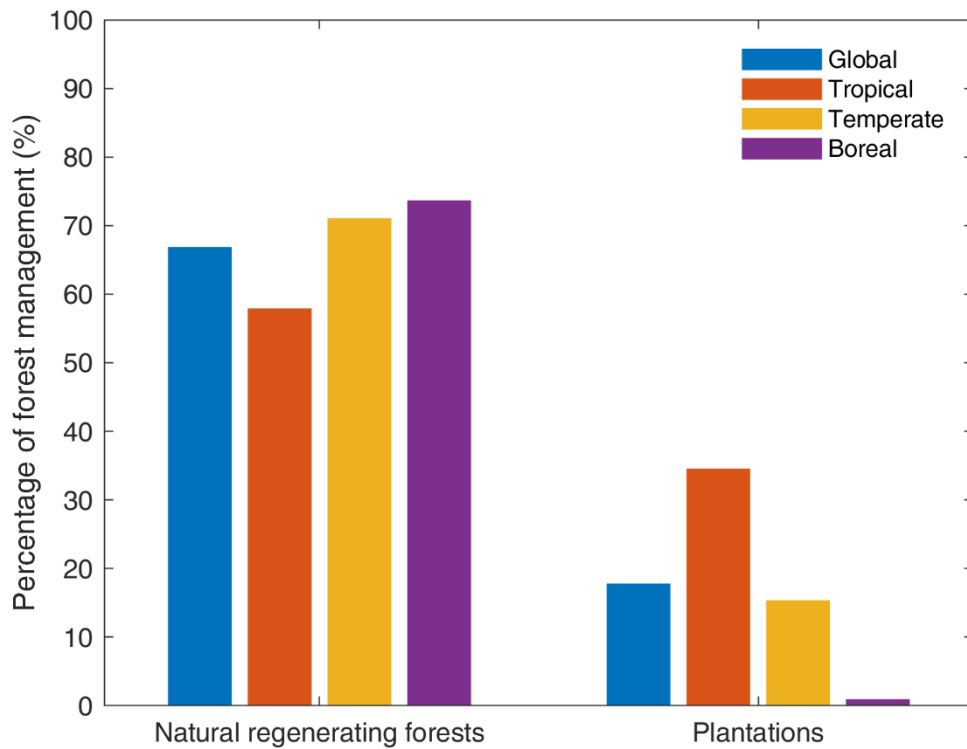
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914 **Figure 3. Drivers of mountain forest loss.** (A) Comparison across all mountains (global), and
 915 in tropical, temperate, and boreal regions. (B) Comparison between the biodiversity hotspots
 916 based on range-size rarity for threatened species (RSR) and inside protected areas in the
 917 hotspots (RSR (PAs)).

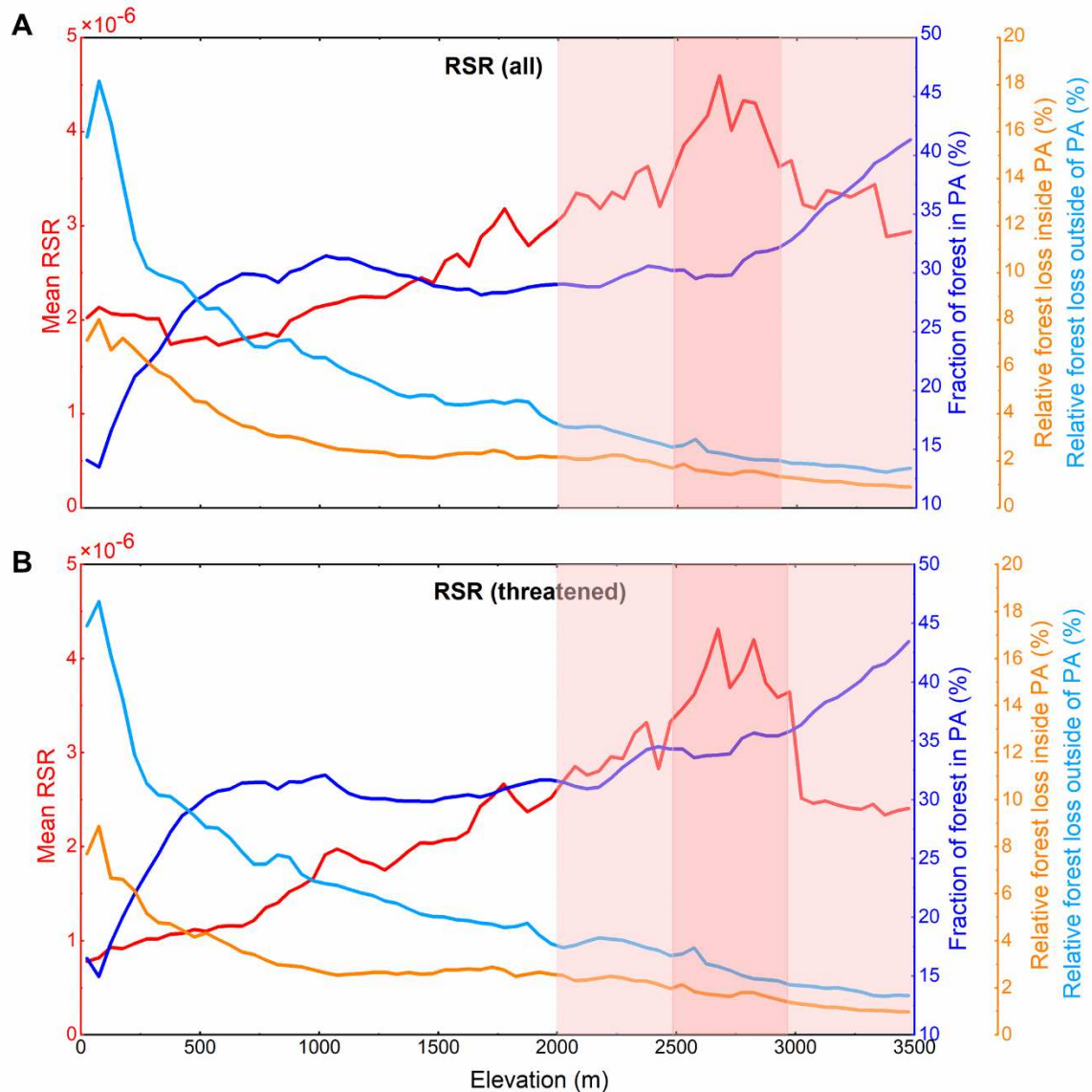
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920 **Figure 4. Proportion of natural regenerating forests and plantations accounting for**
 921 **mountain forest loss.** Naturally regenerating forests include those without any signs of
 922 management (primary forests) and with signs of management (e.g., logging, clear cuts, etc.).
 923 Plantations include planted forests, plantation forests (rotation time up to 15 years), oil palm
 924 plantations and agroforestry.

925



926

927 **Figure 5. Elevational gradients of biodiversity value, protected area (PA) coverage, and**

928 **mountain forest loss inside and outside of PAs within biodiversity hotspots.** Biodiversity

929 hotspots are based on range-size rarity (RSR) for all species (A) and threatened species (B).

930 Mean RSR (red lines) is mean value of biodiversity metric of RSR at each elevation bin on the

931 pixel of 30 m. PA coverage (fraction of forest in PAs) is the ratio of mountain forest within PAs

932 in hotspots versus mountain forest in the corresponding hotspots. Relative forest loss is percent

933 forest loss relative to forest cover in the baseline year 2000. Relative forest loss inside PAs and

934 outside of PAs within hotspots are shown in orange and light blue lines, respectively. The

935 background shading highlights occurrence of the highest levels of biodiversity (light and dark

936 red).

937

938 **Supplemental Data S1. Visual interpretation of Landsat imagery at 5,000 forest loss pixels**
939 **randomly selected from Hansen GFC product across all mountain regions.**